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CRITICAL REVIEW OF THE USE OF BIOCONCENTRATION FACTORS FOR HAZARD CLASSIFICATION OF METALS AND METAL COMPOUNDS

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INTRODUCTION

Bioaccumulation potential is currently used in Europe and North America as a criterion for the hazard classification of organics and is proposed for use with metals. Classification of substances is based upon the premise that hazard can be identified using "inherent" properties of the substance. It is known that organisms bioaccumulate and store metals in their tissues to levels higher than those in their aqueous environment since many of the metals are essential to biological functions. Based on this background, an in-depth review of the bioconcentration literature (laboratory tests) was undertaken to address the following questions: (1) do the scientific data support that metals and metal compounds are bioaccumulative and can this determination be made using the properties of the substance?; (2) do bioconcentration factors for metals and metal compounds provide an indication of the potential for long-term effects in aquatic organisms?; and (3) is it appropriate to use bioaccumulation as a hazard assessment tool for metals and metal compounds?

BACKGROUND

In hazard assessment, bioaccumulation potential is typically assessed using bioconcentration factors (BCFs), on the basis that the ratio of the tissue to water concentration of a chemical is predictive of adverse effects and may reflect a concern for trophic transfer of the substance. BCFs are often used in place of bioaccumulation factors (BAFs) because the latter are not typically available. This report demonstrates that metal BCFs for many taxonomic aquatic groups, and fish in particular, are inversely related to the metal concentration in water. As a result, individual BCFs are not indicative of a metal's bioaccumulation potential in these organisms (i.e., large BCFs do not indicate that bioaccumulation potential is higher; they reflect lower exposure concentrations). Other types of organisms are net accumulators of metals (i.e., they are able to store large quantities of metals in detoxified forms). In these organisms, the BCF may be more indicative of bioaccumulation potential, but the higher bioaccumulation potential does not appear to be related to increased toxicity potential because the metal appears to be stored in a detoxified form. As such, bioaccumulation potential of metals and metal compounds, whether measured as a BCF or tissue residue, cannot be directly correlated with hazard.

METAL ESSENTIALITY

It is well known that a variety of metals are essential for various biological functions, such as enzymatic and metabolic reactions. Metal bioaccumulation is an important process whereby aquatic organisms obtain these essential metals. Aquatic biota regulate their internal concentrations of essential metals in three ways: active regulation, storage, or a combination of active regulation and storage. Active regulators are organisms that maintain stable tissue concentrations by excreting metal at rates comparable to the intake rate. Other biota store metals in detoxified forms, such as in inorganic granules or bound to metallothioneins. Some organisms use a combined regulatory strategy. It should also be noted that non-essential metals are often also regulated to varying degrees because the mechanisms for regulating essential metals are not metal-specific.

In general, essential metals such as copper and zinc tend to be actively regulated by organisms such as decapod crustaceans, algae and fish. Conversely, organisms such as bivalve molluscs, barnacles, and aquatic insects tend to store these metals in detoxified forms. Non-essential metals, such as cadmium, are typically stored in detoxified forms and not actively regulated.

RELATIONSHIP BETWEEN WATER CONCENTRATIONS AND METAL BCFS

As summarized above, many aquatic organisms regulate metals to varying degrees. Consequently, an inverse relationship between water concentrations of metals and the corresponding BCF is often observed. This is because at low water concentrations organisms are actively accumulating essential metals (and often other metals via the same uptake mechanisms) to meet their metabolic requirements. At higher water concentrations, organisms with active regulatory mechanisms are able to excrete excess metals or limit uptake. As a result, metal concentrations in tissue based on a range of exposure concentrations may be quite similar, but the BCFs will be quite variable (i.e., higher BCFs at lower exposure concentrations and lower BCFs at higher exposure concentrations). Consequently, an individual BCF provides little information on the bioaccumulation potential of a metal.

BIOACCUMULATION AS AN INDICATOR OF CHRONIC TOXICITY FOR METALS

The concept that BCFs can be used as an indicator of long-term or chronic toxicity to aquatic organisms stems from the assumption that larger BCFs are indicative of higher tissue concentrations, which in turn result in direct or secondary poisoning. This concept is primarily relevant to organic chemicals with narcosis as the mode of toxic action. However, this relationship does not apply to all chemicals, including metals. In fact, some studies have shown that accumulated metal (whole body residue) may be poorly, or even negatively, correlated with toxicity. Organisms that tend to bioaccumulate metals to high levels (e.g., bivalves, barnacles) do so because they are able to store the metals in detoxified forms (i.e., in granules, bound to metallothionein). Consequently, the magnitude of a metal's BCF cannot be used as a predictor of chronic toxicity.

SECONDARY POISONING AND BIOMAGNIFICATION OF METALS

Secondary poisoning results when toxicant concentrations in an organism reach a level that is toxic to the organisms that feed on it. Substances that bioaccumulate or biomagnify in food webs often are considered to have the greatest potential to cause secondary poisoning. It has been reported that the classic concept of biomagnification and food chain poisoning, based primarily on chemicals such as DDT and PCBs, does not hold for metals (naturally occurring organo-metals may be an exception). This may be explained in part by the limited bioavailability of the inorganic forms of metals in food and by the regulation of metals that occurs in both aquatic and terrestrial organisms. A limited amount of site-specific data are available suggesting that some inorganic metal compounds may result in secondary poisoning, but further research is needed on this topic. Regardless, the literature clearly shows that BCFs cannot be used to estimate bioaccumulation and biomagnification potential for metals and metal compounds. Hence, they are not useful descriptors of hazard.

CONCLUSIONS

This report concludes that metal BCFs are not indicative of the potential for direct toxicity, that there is limited evidence that inorganic forms of metals result in secondary poisoning, and that inorganic forms of metals do not biomagnify in food webs. Consequently, we conclude that bioaccumulation is not an appropriate parameter for assessing the hazard potential of metals.

1. INTRODUCTION

Hazard identification is a process for determining whether chemical substances should be classified as dangerous to the environment. Formal classification protocols based on hazard identification have been established in Europe and an internationally harmonized system for chemical classification is under development through the Organization for Economic Cooperation and Development (OECD). Chemical substances are classified under these protocols based on their persistence, toxicity, and bioaccumulation potential (e.g., EU 1967, 1991). These protocols were primarily derived for organic chemicals.

Over the past several years, a series of workshops have been held to discuss the applicability of these protocols to metals and metal compounds (e.g., OECD 1995, Canada/EU 1996). Based on the workshops and subsequent discussions, it was concluded that the current use of bioaccumulation data in classifying organic compounds is not appropriate for classifying metals and metal compounds. Reasons for reaching this conclusion include:

1.) Unlike organic compounds of anthropogenic origin, metals occur naturally in the environment and many metals are essential nutrients for organisms. As a result, organisms have developed homeostatic processes for regulating internal tissue concentrations of many metals;

2.) Due to these homeostatic processes, an inverse relationship is often observed between bioaccumulation and metal exposure. Consequently, use of a single bioconcentration factor (BCF) for hazard classification is not appropriate for metals;

3.) Bioaccumulation has been used in hazard classification as a surrogate for chronic toxicity. The premise behind this approach is that highly bioaccumulative chemicals are also more likely to cause sublethal/chronic effects. As discussed in this report, the applicability of this concept is not appropriate for metals and metal compounds; and

4.) There is not a clear relationship between bioaccumulation potential and secondary poisoning from metals (e.g., effects via prey ingestion).

The purpose of this report is to provide a detailed review of the scientific literature on each of the above points. The report focuses on the bioaccumulation potential of metals and inorganic metal compounds. Organometallic compounds behave differently in the environment and within organisms. As such, these compounds need to be evaluated separately from a scientific perspective, as well as from a regulatory perspective with regards to hazard classification. In the remainder of this report, the term "metals" refers to metallic elements or inorganic metal compounds (e.g., $CdCl_2$) unless specified otherwise. This report specifically provides detailed summaries and/or analyses on:

1.) The state-of-the-science regarding bioaccumulation of trace metals in aquatic organisms (e.g., Ag, Cd, Cu, Ni, Pb, Zn);

2.) Limitations on the use of BCFs for hazard identification;

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3.) Appropriate use of bioaccumulation data for metals as an indicator of secondary poisoning (Canada/EU 1996); and

4.) Evaluation of biomagnification (increased concentrations at successively higher trophic levels) for metals.

2. REVIEW OF METAL BIOACCUMULATION

Although bioaccumulation data for metals and metal compounds can be an extremely useful tool in various applications when used appropriately, these data are inappropriate for hazard classification of metals and most metal compounds. Metal bioaccumulation is an important process whereby aquatic organisms acquire essential elements for key metabolic processes. The essentiality of certain metals has led organisms to develop strategies for actively regulating and/or sequestering metals in detoxified forms. Consequently, bioaccumulation potential, and the parameters typically used to estimate it (e.g., BCFs), should not be interpreted for metals in the same manner as for synthetic organic compounds. The more important issue, as Beyer (1986) pointed out, is that overemphasizing bioaccumulation or biomagnification potential often diverts from the more important question of whether metal concentrations in the environment are toxic to humans, wildlife, or aquatic life.

The following first provides a brief overview of the strategies used by aquatic biota to regulate both essential and non-essential metals. Bioconcentration factors for a variety of metals and aquatic organisms are then graphically presented to demonstrate the difficulties in interpreting a single BCF. Finally, a review of the scientific literature is presented to provide a summary of available information on metal bioaccumulation and to evaluate the potential to use bioaccumulation data as an indicator of direct toxicity or toxicity via secondary poisoning.

2.1 METAL ESSENTIALITY

It is well demonstrated that a number of metals are essential for various biological functions and are critical in many of the enzymatic and metabolic reactions occurring within an organism. Several metals (e.g., sodium, potassium, magnesium, and calcium) occur in large concentrations in organisms. A second set of metals, termed trace metals, occur at much lower concentrations (normally <0.01%) in organisms (Simkiss and Taylor 1989, Venugopal and Luckey 1974). Trace metals can generally be categorized into essential and non-essential groups. Simply defined, essential metals are those necessary for tissue metabolism and growth (Leland and Kuwabara 1985). Essential trace metals and some of the roles they play in organism metabolism and growth are summarized in Table 1. Note that not all of these metals are known to be essential to aquatic biota. For example, chromium is known to be essential to terrestrial vertebrates, but no references were found which identified chromium as essential to aquatic life.

Other trace metals, such as cadmium, lead, mercury and silver, are generally considered non-essential. However, these metals also appear to be regulated to varying degrees because the mechanisms for regulating essential metals (described below) are not metal-specific (Phillips and Rainbow 1989).

Table 1. Examples of some essential trace metals and their function.

Metal	Example of function
Chromium	Cofactor for insulin action
Cobalt	Component of vitamin B_{12}
Copper	Prosthetic group of cytochrome and hemocyanin
Iron	Prosthetic group of hemoglobin
Manganese	Cofactor of arginase
Molybdenum	Cofactor of xanthine oxidase
Nickel	Cofactor of urease
Selenium	Cofactor of glutathione peroxidase
Zinc	Carbonic anhydrase, carboxy-peptidase A and B

From: Depledge and Rainbow (1990), Parametrix (1995), and Goyer (1996).

2.2 HOMEOSTATIC CONTROL OF METALS

Given the number and importance of trace metals to aquatic life, organisms have developed a variety of homeostatic control mechanisms to regulate their concentrations *in vivo*. The mechanisms by which metal concentrations are regulated vary widely between organisms (George et al. 1980, Mason and Nott 1981, Rainbow et al. 1980, Simkiss 1981, White and Rainbow 1984, Rainbow 1988, Viarengo 1989, Depledge and Rainbow 1990) and, as such, aquatic organisms are generally classified as regulators, partial regulators, or non-regulators (Phillips and Rainbow 1989). These terms are somewhat of a misnomer as all organisms regulate metals – it is the mechanisms by which they do so that distinguishes them. Consequently, we have renamed these categories to more accurately describe the mechanisms by which metals are regulated.

Active Regulation: Active regulators are organisms that maintain stable tissue concentrations by excreting metal at rates comparable to the intake rate. Some decapod crustaceans, for example, regulate zinc and copper using this mechanism (Rainbow 1988).

Active Regulation/Storage: Organisms in this group control internal metal concentrations through a combination of active regulation and storage. Trace metals are usually stored in the form of metallothioneins and occasionally as granules at high ambient concentrations (Phillips and Rainbow 1989). Storage is normally in the hepatopancreas and kidney. Metals stored in this fashion are generally metabolically available. Fish and many invertebrates use this combined strategy of regulation and storage.

Storage: Some organisms store large concentrations of metals in a detoxified, normally granular, form. Storage location varies by metal and by species (Phillips and Rainbow 1989). For any given species, organisms may exclusively use storage for one metal while using an active regulation/storage strategy for a different metal. Additionally, some bivalve molluscs use both metallothionein and granular storage mechanisms for detoxification. Metal and species-specific examples of granular storage include zinc for barnacles (Rainbow 1987) and copper for oysters (Brown 1982).

The following provides a review, by metal, of the regulatory strategies used by different aquatic biota. Most research on this topic has focused on cadmium, copper, and zinc. As a result, most of the examples provided below are based on these metals. Where available, examples of regulatory strategies are also provided for other metals. It should be noted that the regulatory mechanisms used by aquatic biota to regulate metals do not always fall into the three discrete categories defined above, but fall along a gradient of strategies (Rainbow et al. 1990). A summary of the mechanisms used by different biota for various metals is provided in Table 2.

Table 2. Examples of regulatory mechanisms used by aquatic organisms for several metals.

Metal	Active Regulation	Active Regulation/Storage	Storage
Cadmium			
Bivalves			X
Gastropods			
Annelids		X	
Insects			X
Amphipods			
Decapods		X	X
Barnacles			X
Fish		X	
Copper			
Bivalves			X
Gastropods			X
Annelids		X	
Insects			X
Amphipods		X	X
Decapods	X	X	
Barnacles			X
Fish	X		
Lead			
Bivalves			X
Gastropods			X
Annelids		X	
Insects			X
Amphipods			X
Decapods		X	X
Barnacles			X
Fish		X	
Nickel			
Bivalves			
Gastropods			
Annelids			
Insects			
Amphipods			
Decapods			
Barnacles			
Fish			
Silver			
Bivalves			
Gastropods			X

Table 2. Examples of regulatory mechanisms used by aquatic organisms for several metals. (Continued)

Metal	Active Regulation	Active Regulation/Storage	Storage
Annelids			X
Insects			
Amphipods			
Decapods			
Barnacles			
Fish			
Zinc			
Bivalves		X	X
Gastropods		X	X
Annelids	X		
Insects			X
Amphipods		X	X
Decapods	X		
Barnacles			X
Fish	X		

2.2.1 Cadmium

Active Regulation: There is no evidence in the scientific literature that non-essential metals such as cadmium are actively regulated by aquatic biota (Rainbow 1996).

Active Regulation/Storage: Again, since no aquatic organisms are known to actively regulate cadmium, there are no examples of species that use this combined strategy. Using growth dilution, decapods are able to regulate low cadmium concentrations in a manner that approaches active regulation, but in reality is not (Rainbow et al. 1990). Accumulated cadmium in decapods is usually associated with metallothioneins and granules which may represent detoxified forms. In natural systems, where cadmium concentrations in surface waters are very low, the growth rates of organisms can dilute the cadmium concentrations in decapods. When bioavailable cadmium concentrations reach a high enough level, the detoxification system fails and mortality results.

Storage: Barnacles are common examples of organisms that store metals in detoxified forms. Rainbow et al. (1980), for example, demonstrated that cadmium binds to low and high molecular weight proteins in the barnacle *Semibalanus balanoides*. In shrimp, cadmium can be tolerated at concentrations significantly above 'normal' tissue levels. Because cadmium is known to be an analogue for zinc in many metalloproteins, it is possible that cadmium only becomes toxic when competing strongly with zinc for binding sites (White and Rainbow 1982). Cadmium also seems to accumulate proportionally to its exposure level in the marine isopod *Idotea baltica* and is stored in the organisms as granules in the hepatopancreas (de Nicola et al. 1993).

The scallop *Mizuhopecten yessoensis* is also a net accumulator (i.e., the uptake rate exceeds the excretion rate) of cadmium (Lukyanova et al. 1993). Cadmium was observed to accumulate to high levels in the kidney and hepatopancreas of the scallop in an age-dependent manner, even at relatively low environmental concentrations. It appears that cadmium in the scallop binds to high molecular weight proteins that have the same biological significance as metallothioneins in other animals.

Kaland et al. (1993) studied the accumulation of cadmium in the marine gastropod *Nassarius reticulatus*. Cadmium was sequestered by a minor pool of high molecular weight proteins and a major pool of proteins with molecular weights similar in size to mammalian metallothionein. In exposed organisms, cadmium was found to also bind to very low molecular weight proteins. The authors suggest that the cadmium in this pool represents the "spill over" from the detoxified cadmium bound to metallothionein-like proteins and is more metabolically available.

Spehar et al. (1978) exposed insects (*Pteronarcys dorsata* and *Hydropsyche betteni*) and a snail (*Physa integra*) to cadmium. All species were shown to be net accumulators of cadmium. Cadmium residues in *P. dorsata* and *P. integra* increased with increasing exposure concentration, while residues in *H. betteni* reached an equilibrium at higher exposure concentrations.

2.2.2 Copper

Active Regulation: Rainbow and White (1989) determined that the shrimp *Palaemon elegans* can actively regulate aqueous dissolved copper concentrations up to 100 μg/L. As summarized in Rainbow and White (1989), other decapods known to regulate their body concentrations of copper include the lobster *Homarus gammarus*, the crab *Carcinus maenas*, and the shrimp *Crangon crangon*. The amphipod *Echinogammarus pirloti* uses a different mechanism for actively regulating internal copper concentrations. *E. pirloti* does not actively excrete excess copper, rather, it accumulates copper at a low net rate relative to its body growth rate (Rainbow and White 1989).

It appears that some species of polychaetes may also be able to actively regulate body concentrations of copper (Young et al. 1979, Pesch and Morgan 1978). Young et al. (1979) observed that copper residues in the polychaete *Eudistylia vancouveri* were relatively constant over a 33 day exposure period and that only in the highest exposure concentration did the residue concentration steadily rise during the exposure period. The authors hypothesized that the copper residues in the polychaetes were in equilibrium with the lower treatment mediums, but that regulatory abilities were exceeded in the highest copper concentration. Pesch and Morgan (1978) also suggest that the polychaete *Neanthes arenaceodentata* may be able to actively excrete excess copper from its body.

Active Regulation/Storage: As discussed above, many decapods are active regulators of copper. In regulating internal copper concentrations, copper is bound to metallothioneins, a process which can be of significance in detoxification (Rainbow et al. 1990). For example, after breakdown of active copper regulation, *P. elegans* can survive with accumulated copper concentrations up to approximately 700 mg/kg, suggesting that at least some of the accumulated copper is in detoxified form (Rainbow et al. 1990). In addition, copper-rich granules may be present in hepatopancreatic cells at these high copper concentrations (Rainbow et al. 1990). Consequently, although decapods actively regulate internal copper concentrations, they also have the capacity to store some excess copper in detoxified forms.

Storage: Like cadmium, copper binds to low and high molecular weight proteins in the barnacle *Semibalanus balanoides* (Rainbow et al. 1980). Another example of net accumulators are chironomid larvae (Timmermans and Walker 1989).

The marine gastropod (*N. reticulatus*) has been observed to accumulate copper in all tissues – but given that the gastropods were exposed to low copper concentrations it is not possible to determine their regulatory strategy because their regulatory strategy may not have been saturated (Kaland et al. 1993). Most copper was bound to proteins similar to mammalian metallothionein. In addition, Brown (1982) observed that oysters store copper in granules.

2.2.3 Lead

Active Regulation: No studies were identified in the scientific literature demonstrating that lead tissue concentrations can be actively regulated by aquatic biota. This is expected since lead is a non-essential metal and there is no evidence for active regulation of non-essential metals.

Active Regulation/Storage: No data were identified on organisms that use these combined regulatory mechanisms.

Storage: Lead will bind to metallothionein, but also has an affinity (probably higher) for other metabolic ligands as it is, often associated with deposited inorganic granules with high concentrations of calcium (Rainbow 1988). Hopkin and Nott (1979) demonstrated that the shore crab (*Carcinus maenas*) detoxifies lead in calciferous granules in the midgut gland. The midgut gland connects to the alimentary tract, where these granules have the potential to be lost through defecation.

Because lead BCF data for bivalves (discussed later in this report) tend to remain constant regardless of exposure concentration, it is likely that lead is stored in many of these organisms. This has been suggested for the zebra mussel *Dreissena polymorpha* (Bleeker et al. 1992, Kraak et al. 1994), the blue mussel *Mytilus edulis* (Talbot et al. 1976, Schulz-Baldes 1974), the Eastern oyster *Crassostrea virginica* (Pringle et al. 1968, Shuster and Pringle 1969, Zaroogian et al. 1979), and the soft-shell clam *Mya arenaria* (Pringle et al. 1968). Spehar et al. (1978) exposed insects (*Pteronarcys dorsata* and *Brachycentrus* sp.), a snail (*Physa integra*), and an amphipod (*Gammarus pseudolimnaeus*) to lead. All species were shown to be net accumulators of lead, with residues increasing with increasing exposure concentration.

2.2.4 Nickel

No studies were identified on the mechanisms by which aquatic biota regulate nickel. Given that it is an essential metal, and that it is a divalent metal, it is likely that the mechanisms used are similar to those for copper or zinc. That is, some organisms actively regulate nickel while others sequester it to varying degrees.

2.2.5 Silver

Active Regulation: No examples of organisms that use this method of regulation for silver were identified in the scientific literature.

Active Regulation/Storage: No examples of organisms that use this method of regulation for silver were identified in the scientific literature.

Storage: The data are very limited, but it appears that silver concentrations in the polychaete worm *Neries diversicolor* tend to be directly related to concentrations in sediments (Bryan 1979). This suggests that silver is stored in these organisms. It also appears that silver is not actively regulated in the gastropod *Littorina littorea* (tissue concentrations in the gastropod increased proportionally to the silver concentration in food items) (Bryan 1979).

2.2.6 Zinc

Active Regulation: Like copper, Rainbow and White (1989) determined that the shrimp *Palaemon elegans* can actively regulate aqueous dissolved zinc concentrations up to 100 µg/L. Another closely related marine shrimp (*Palaemon serratus*) and the freshwater decapod *Austropotamobius pallipes* also appears to be active regulators of zinc (Devineau and Amiard Triquet 1985, Rainbow and Dallinger 1993). As summarized by Rainbow and White (1989), there is evidence that a variety of other decapods actively regulate their body concentrations of zinc, including lobster (*Homarus gammarus*), crab (*Carcinus maenas*, *Maia squinado*), and shrimp (*Crangon crangon*).

There is also evidence to suggest that the freshwater oligochaete *Lumbriculus variegatus* may actively regulate zinc, and the estuarine polychaete *Neries diversicolor* is also known to actively regulate it (Rainbow and Dallinger 1993). Also like copper, the amphipod *E. pirloti* does not actively excrete excess zinc, but takes it up at a low net rate relative to its body growth rate (Rainbow and White 1989).

Active Regulation/Storage: The marine mussel (*Mytilus edulis*) excretes much of the zinc that it accumulates from the kidney (George and Pirie 1980). Freshwater mussels, including *Dreissena polymorpha*, *Unio pictorum*, and *Velesunio ambiguus* also appear to regulate tissue levels of zinc (Kraak et al. 1993, Rainbow and Dallinger 1993). Accordingly, mussels are weak accumulators (or active regulators/storers) of zinc (Rainbow 1993).

The gastropod *N. reticulatus* may be able to actively regulate zinc concentrations because only minor increases in the body content of zinc were observed at very high zinc concentrations (Kaland et al. 1993). Most zinc was found to be associated with very low molecular weight proteins, although small amounts were also bound to high molecular weight components. The authors state it is unlikely that substantial amounts of zinc would be associated with intracellular granules in this gastropod.

Storage: Barnacles store very large concentrations of accumulated zinc in the form of apparently detoxified zinc phosphate granules (Rainbow and White 1989). Rainbow et al. (1980) also demonstrated that zinc binds to low and high molecular weight proteins in the barnacle *Semibalanus balanoides* (Rainbow et al. 1980). The barnacle *Elminius modestus* accumulates and stores zinc in a detoxified granular form, and therefore, the zinc in the barnacle increases with increasing zinc exposure (Rainbow and White 1989). In the Thames estuary, U.K., zinc concentrations have been measured as high as 150,000 mg/kg in the barnacle *Balanus improvisus* with, again, zinc being stored in granules in the form of detoxified zinc pyrophosphate (Rainbow et al. 1990).

Oysters (*Ostrea edulis*) accumulate high concentrations of zinc in detoxified granules (George et al. 1978). Accordingly, oysters are considered strong accumulators of zinc (Rainbow 1993).

Timmermans and Walker (1989) reported there was no evidence that the chironomids (midges) *Chironomus riparius* and *Stictochironomus histrio* actively regulate their zinc body burdens. Body burdens increased with increasing zinc exposure, but zinc was lost with each cast exuvium.

2.2.7 Summary

Although control mechanisms have evolved largely for essential metals, the mechanisms also operate quite successfully for many non-essential metals as well, allowing organisms to sequester, for example cadmium, mercury, and silver via metallothioneins (Viarengo 1989), cadmium via high and low molecular weight proteins (Rainbow et al. 1980), and lead in granules (Phillips and Rainbow 1989). However, there do not appear to be any examples in the literature of active regulation of non-essential trace metals (Rainbow 1996).

The key point regarding the regulation of essential metals is that bioaccumulation by organisms is an intrinsic property of these metals and is essential for life. Control mechanisms for metal bioaccumulation are fundamentally different from organic chemicals of anthropogenic origin. Hence, the application of bioaccumulation-based hazard classification criteria developed for organics appear inappropriate for use with metals. The following section provides a critical review of the bioaccumulation data for aquatic biota and demonstrates that individual BCFs for metals are not indicative of hazard potential.

2.3 RELATIONSHIP BETWEEN WATER CONCENTRATIONS AND BCFS

Many organisms can regulate metals to varying degrees. As a result, an inverse relationship between water concentrations of metals and the corresponding BCF is often observed. This relationship exists because at low water concentrations organisms are actively accumulating essential metals (and often other metals via the same uptake mechanisms) to meet their metabolic requirements. At higher water concentrations, organisms with active regulatory mechanisms are able to excrete excess metals or limit uptake. Consequently, the metal concentration in the tissue(s) of such an organism may be the same regardless of the water concentrations to which it was exposed. Despite the tissue concentrations remaining the same (or nearly the same), the BCF at the low water concentration may be quite high (suggesting high bioaccumulation potential) and the BCF at the high water concentration may be quite low (suggesting low bioaccumulation potential). In reality, the bioaccumulation potential of the metal expressed as actual body burden is the same under each scenario. To better quantify and demonstrate this relationship for different metals and organisms, a search of the scientific literature was conducted to summarize the BCF and water concentration relationship. The following summarizes this review and subsequent analyses.

For the purposes of this analysis, the review of the scientific literature was thorough, but not exhaustive. All studies were critically reviewed for quality, with the most important considerations being whether steady-state tissue concentrations were achieved in the test and whether metal concentrations were measured over the duration of the exposure period. Following U.S. EPA guidelines (Stephan et al. 1985), it was assumed that 28 days was of sufficient duration for steady state to be reached in fish (if it was not clear whether steady state was reached by the end of the test). In addition, BCF data based on exposure concentrations resulting in significant effects to the exposed organisms were not used.

The following discusses the BCF data for six representative metals: cadmium, copper, lead, nickel, silver, and zinc. These metals were selected because their BCF databases are relatively large, covering a variety of species. In addition, these six metals represent both essential (copper, nickel, zinc) and non-essential metals (cadmium, lead, silver). Ranges of BCFs for different taxonomic groups are provided, as well as an evaluation of the relationship between water concentrations and BCFs for individual species. To evaluate this relationship, water concentrations and associated BCFs were plotted on a log-log scale. The relationship tends to be linear or very near linear, so linear regressions for fish species were compared to those for invertebrate species. In addition, regressions for different taxa groups (e.g., bivalves versus non-bivalves) were evaluated. If an organism actively regulates a metal, the slope of this relationship is expected to be near negative one, while the slope is expected to be near zero in an organism that stores a metal in proportion to the metal concentration in water. The slope is expected to be somewhere between negative one and zero for organisms that use a combined strategy of active regulation and storage. With the exception of algae BCF data, the results of these analyses are presented and discussed for each metal below. Algae BCF data are limited for most metals, so these BCFs are first discussed as a group.

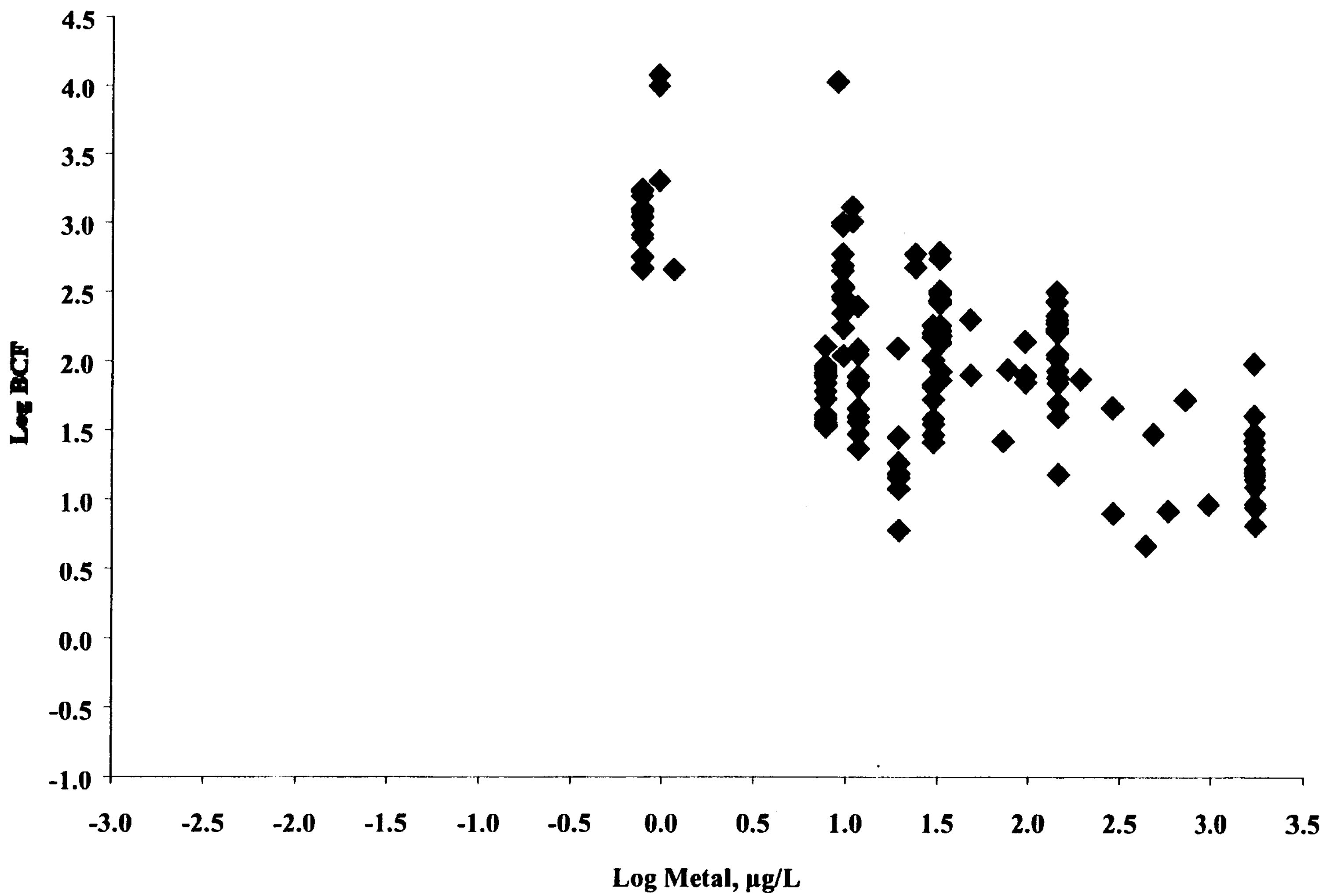
For the six metals evaluated, algae BCF data are based on a range of exposure concentrations were typically limited to just one species. Accordingly, the BCF data across all metals were combined into a single figure (Figure 1). There is a fair amount of scatter in the data, but this is to be expected given that data were pooled for multiple metals and species, and from tests conducted in multiple laboratories using multiple test methods. Despite this scatter, there is a clear inverse relationship between the BCFs and exposure concentrations. This suggests that the metals are being regulated by a similar mechanism. The following sections demonstrate that this is a common pattern for a variety of other aquatic biota as well.

2.3.1 Metal-Specific Examples

2.3.1.1 Cadmium

Cadmium BCF data for algae are primarily limited to the diatom *Ditylum brightwellii*. The BCFs for this diatom are quite low, ranging from approximately 5 to 27 (Canterford et al. 1978). Given the continuum of regulatory strategies used by invertebrates, the BCFs for these organisms are highly variable. The largest BCF identified was 33,500 in the grass shrimp *Palaemonetes pugio* (Pesch and Stewart 1980). However, this BCF was derived based on an extremely low water concentration of 0.1 µg/L. At a much higher water concentration of 83 µg/L, the BCF declines drastically to 157. A similar pattern is observed in the BCF data for the amphipod *Hyalella azteca*. At water concentrations of 0.01 and 9.0 µg/L, the BCFs are approximately 30,000 and 512, respectively (Stephenson and Mackie 1989, Borgmann et al. 1991). The BCFs for bivalves range from a low of approximately 10 in the mussel *Elliptio complanata* (Wang and Evans 1993) to approximately 3,000 in the Eastern oyster *Crassostrea virginica* (Zaroogian and Cheer 1976).

Figure 1. Metal BCFs for Algae and Diatoms.



Overall, cadmium BCFs for fish are lower than those for most invertebrates, but again, the magnitude of the BCFs tend to be dependent on exposure concentration. In one rainbow trout (*Oncorhynchus mykiss*) study, for example, BCFs ranged from 12,000 at a water concentration of 0.01 µg/L to 200 at a water concentration of 4.8 µg/L (Kumada et al. 1973).

To further evaluate the inverse relationship between water concentration and BCF, species with BCFs determined over a range of water concentrations were plotted on a log-log scale. Figures 2, 3, and 4 graphically demonstrate that cadmium BCFs for most species tend to decrease with increasing exposure concentration for fish, non-bivalve invertebrates, and bivalves, respectively. The decreasing trends suggest that most fish and invertebrate species have the ability to actively regulate internal cadmium concentrations using a mechanism that appears similar to active regulation.

These figures clearly demonstrate that BCFs cannot be interpreted without consideration of the exposure concentrations. As discussed, the available BCFs for rainbow trout range from approximately 12,000 to 200 between exposure concentrations of 0.01 and 4.8 µg/L. The BCF of 12,000, if considered by itself, suggests the bioaccumulation potential of cadmium is quite high, while the BCF of 200, if considered by itself, suggests the bioaccumulation potential of cadmium is relatively low. Despite the BCFs differing by a factor of 60, the tissue concentrations only differ by a factor of eight. It is more appropriate, therefore, to compare the relative BCFs between species at the same water concentrations or by considering the actual metal residues in the organisms. If the regressions in Figures 2, 3, and 4 are compared, for example, it can be discerned that fish tend to accumulate lower levels of cadmium than invertebrates. Not surprisingly, if the actual metal residues in tissue are considered, organisms that are known to store metals in detoxified forms have the highest tissue residues: bivalves, polychaetes, and insects. The residues measured in fish are two orders of magnitude lower. As discussed below, these relationships are not just relevant to cadmium, but to a variety of other metals as well.

2.3.1.2 Copper

Copper BCFs for algae tend to be lower than for other types of aquatic biota, as observed for cadmium. Copper algal BCFs range from approximately 40 in the diatom *Ditylum brightwellii* (Canterford et al. 1978) to approximately 600 in the alga *Heteromastix longifillis* (Riley and Roth 1971). The BCFs for invertebrates are again highly variable due to the differing regulatory strategies of these organisms. The largest copper BCFs in invertebrates appear to be for the Eastern oyster *Crassostrea virginica*; BCFs are approximately 28,000 and 20,000 at water concentrations of 25 and 50 µg/L, respectively (Shuster and Pringle 1969). The BCFs appear to decline with increasing exposure concentration, but the strength of this relationship is uncertain because the exposure concentrations only span a factor of two. Measured BCFs tend to be lower for non-bivalve invertebrates; they range from 442 to 10,800 in two species of amphipods (Borgmann and Norwood 1995, Ahsanullah and Williams 1991), 320 to 1,040 in isopods (Brown 1977), and 260 to 4,547 in four species of polychaetes (Millanovich et al. 1976, McKlusky and Phillips 1975).

Figure 2. Cadmium BCFs for Fish.

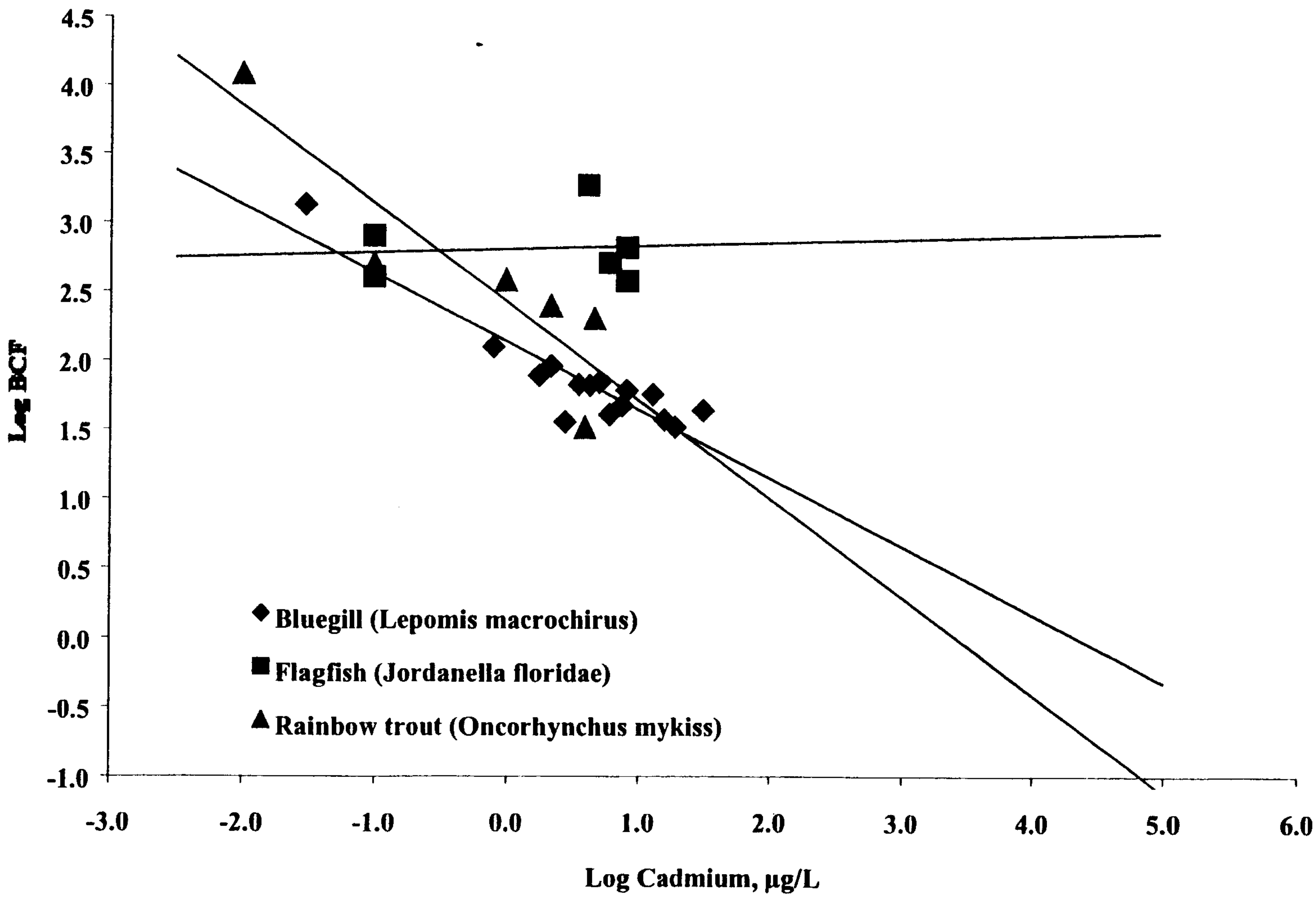


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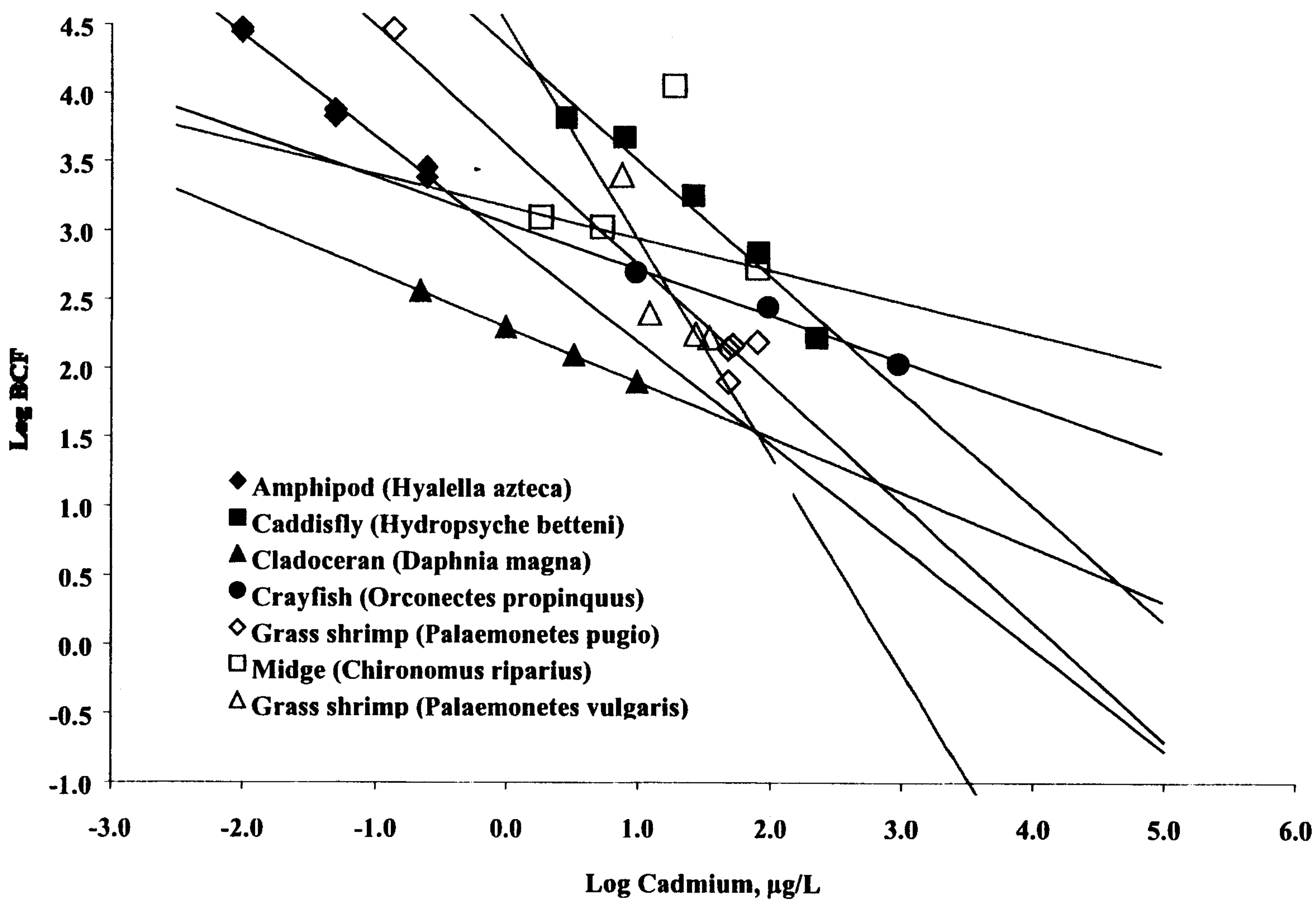
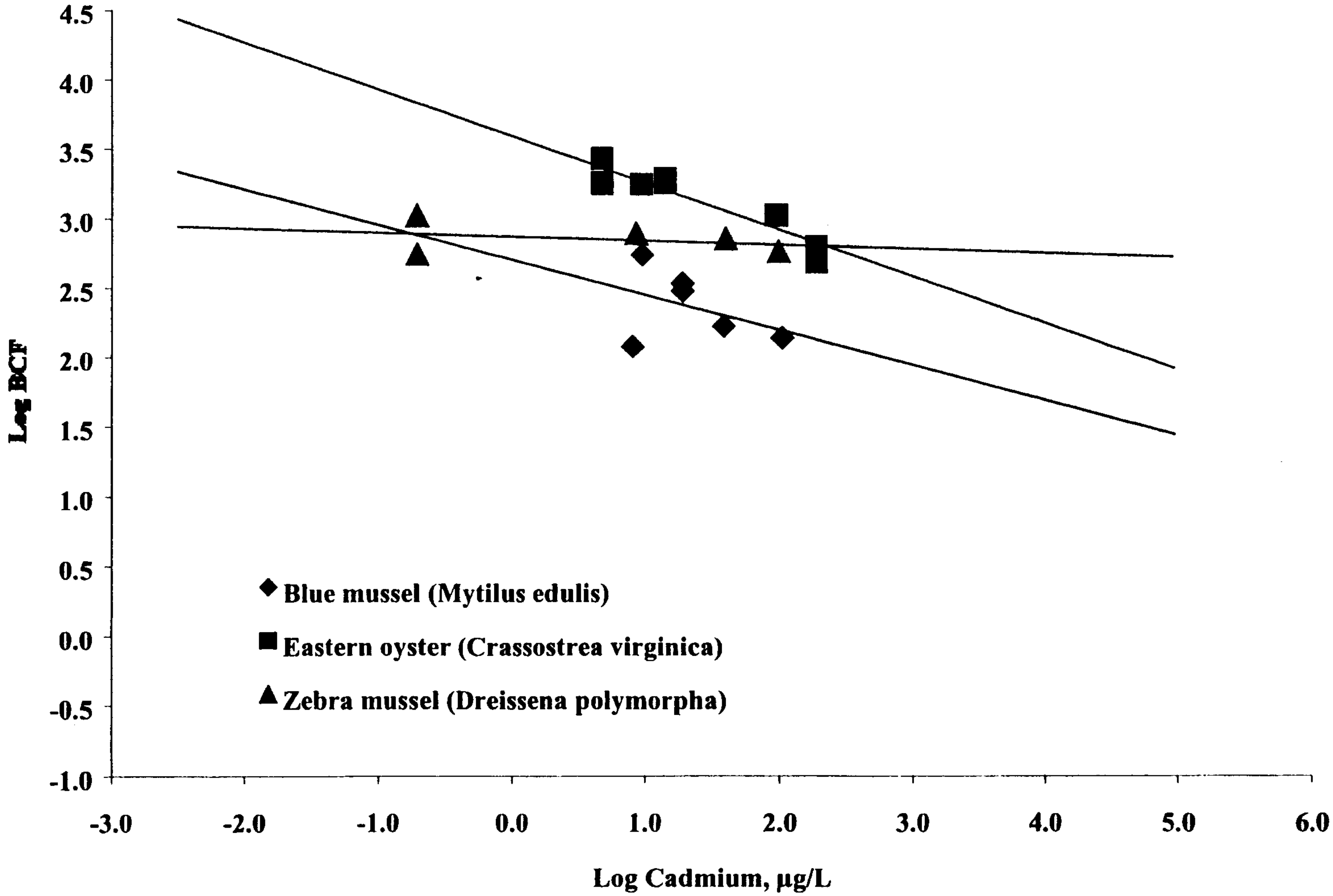
Figure 3. Cadmium BCFs for Non-Bivalve Invertebrates.



AR 044599

Figure 4. Cadmium BCFs for Bivalves.



Copper BCFs were plotted versus their water concentrations to determine whether an inverse relationship also exists for copper. No appropriate whole body copper BCFs were identified for fish, but BCFs for non-bivalve invertebrates (one amphipod and two polychaetes) were determined over a range of exposure concentrations. The BCFs for all of these species were inversely related to water concentration (Figure 5). This observed relationship for polychaetes is not surprising since there is evidence that some species actively regulate their internal copper concentrations (Pesch and Morgan 1978, Young et al. 1979). A similar pattern was observed for three of the four bivalve species with sufficient BCF data, although the slopes are more variable (Figure 6). The relationship for the bay scallop *Argopecten irradians* is positive, although the mechanism for this is unclear. Zaroogian and Johnson (1983) note that weight and spawning are responsible for fluctuations in tissue copper concentrations. It is possible that "normal" variability in growth, and therefore copper concentration, may be responsible for the increasing relationship that was observed (although this was not confirmed). The species with the highest tissue residues were the Eastern oyster, an isopod, and a polychaete.

2.3.1.3 Lead

Lead BCFs in algae are quite variable and tend to be larger than those for cadmium and copper, with BCFs ranging from 26.1 in *Dunaliella tertiolecta* (Riley and Roth 1971) to 14,800 in *Selenastrum capricornutum* (Vighi 1981). The non-bivalve invertebrate with the highest measured BCF (8,000) is the isopod *Asellus meridianus* (Brown 1977). The BCFs for most other non-bivalve invertebrates are generally less than 1,000. The BCFs for bivalves tend to be higher than those for other types of invertebrates, but still less than 5,000. Lead BCFs for fish are limited, but BCFs for brook trout *Salvelinus fontinalis* were less than 100 over a range of water concentrations (Holcombe et al. 1976).

Like cadmium and copper above, BCFs measured over a range of exposure concentrations were plotted for those species with sufficient data. Figure 7 show a decreasing trend between BCFs and water concentration for an amphipod, caddisfly, stonefly, snail, and fish. A wide range of water concentrations were tested for these species, further strengthening this relationship. Conversely, lead BCFs for four species of bivalves were nearly constant over a wide range of exposure concentrations (Figure 8). This is consistent with Phillips and Rainbow's (1989) observation that bivalves tend to store lead in detoxified granules.

2.3.1.4 Nickel

Nickel BCF data are primarily limited to a cladoceran (*Daphnia magna*), three bivalve species, and a fish (*Pimephales promelas*). The cladoceran BCFs are all less than 200 (Hall 1982, U.S. EPA 1986). The measured BCFs for *Cerastoderma edule*, a bivalve, range from 3,198-59,600 (Wilson 1983), but are less than 350 for blue mussels and Eastern oysters (Zaroogian and Johnson 1984). The measured BCFs in fish are all less than 110 (Lind et al. Manuscript). The high BCFs for *C. edule* are a function of both experimental design and the propensity for this species to bioaccumulate nickel. The highest BCF (59,600) is certainly a function of extremely low water nickel concentrations (e.g., 0.1 µg/L) to which they were exposed. However, even at higher water

Figure 5. Copper BCFs for Non-Bivalve Invertebrates.

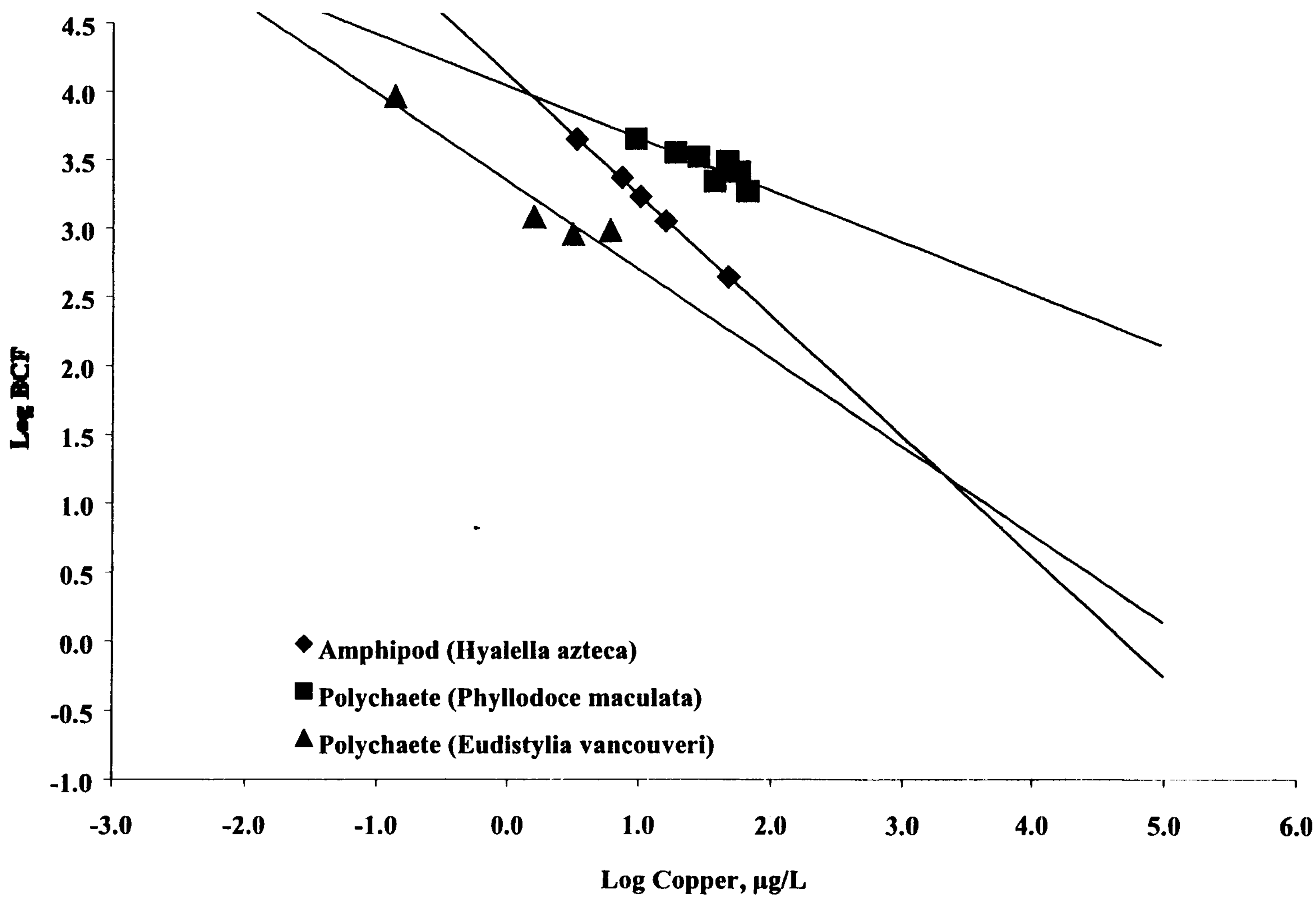


Figure 6. Copper BCFs for Bivalves.

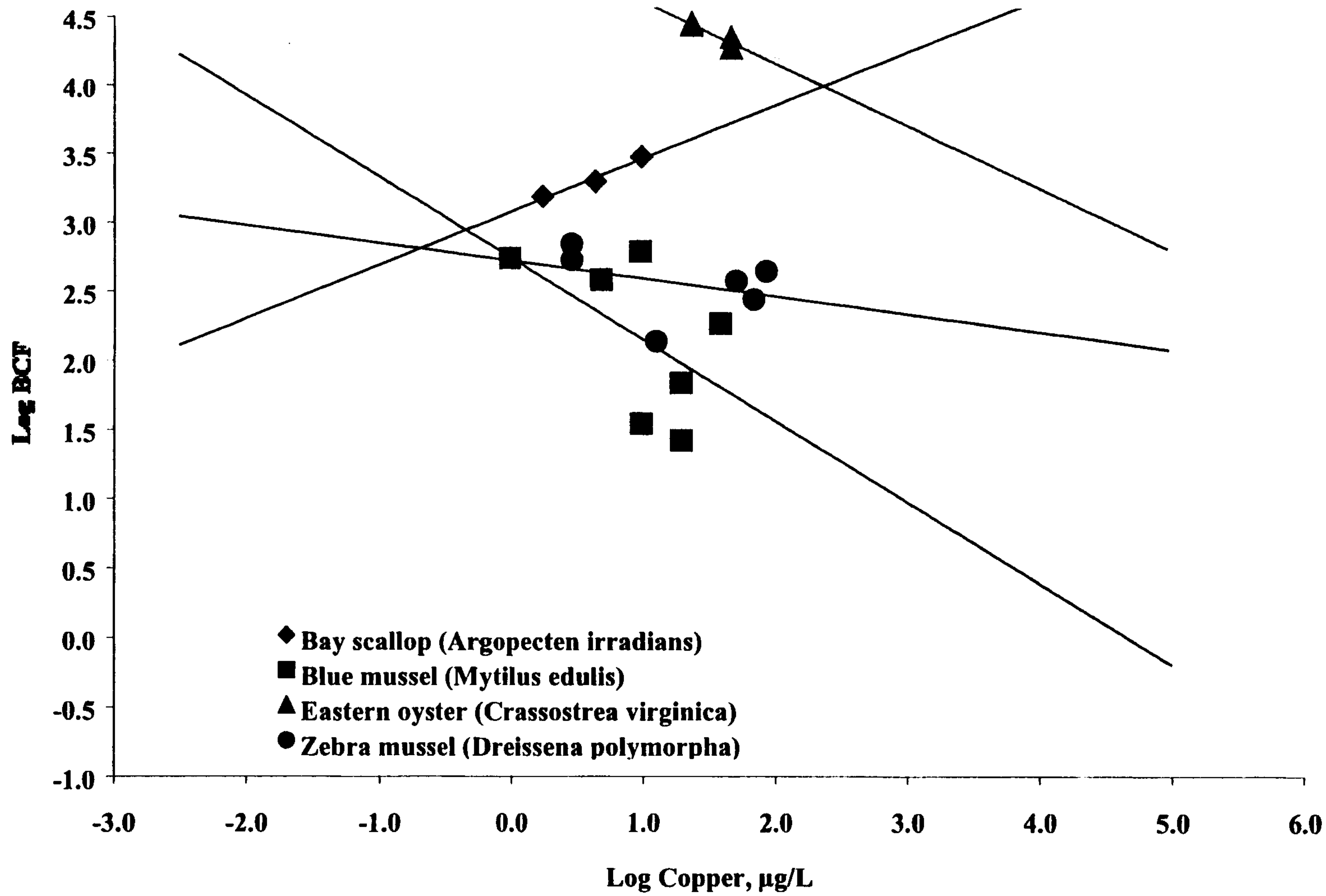


Figure 7. Lead BCFs for Fish and Non-Bivalve Invertebrates.

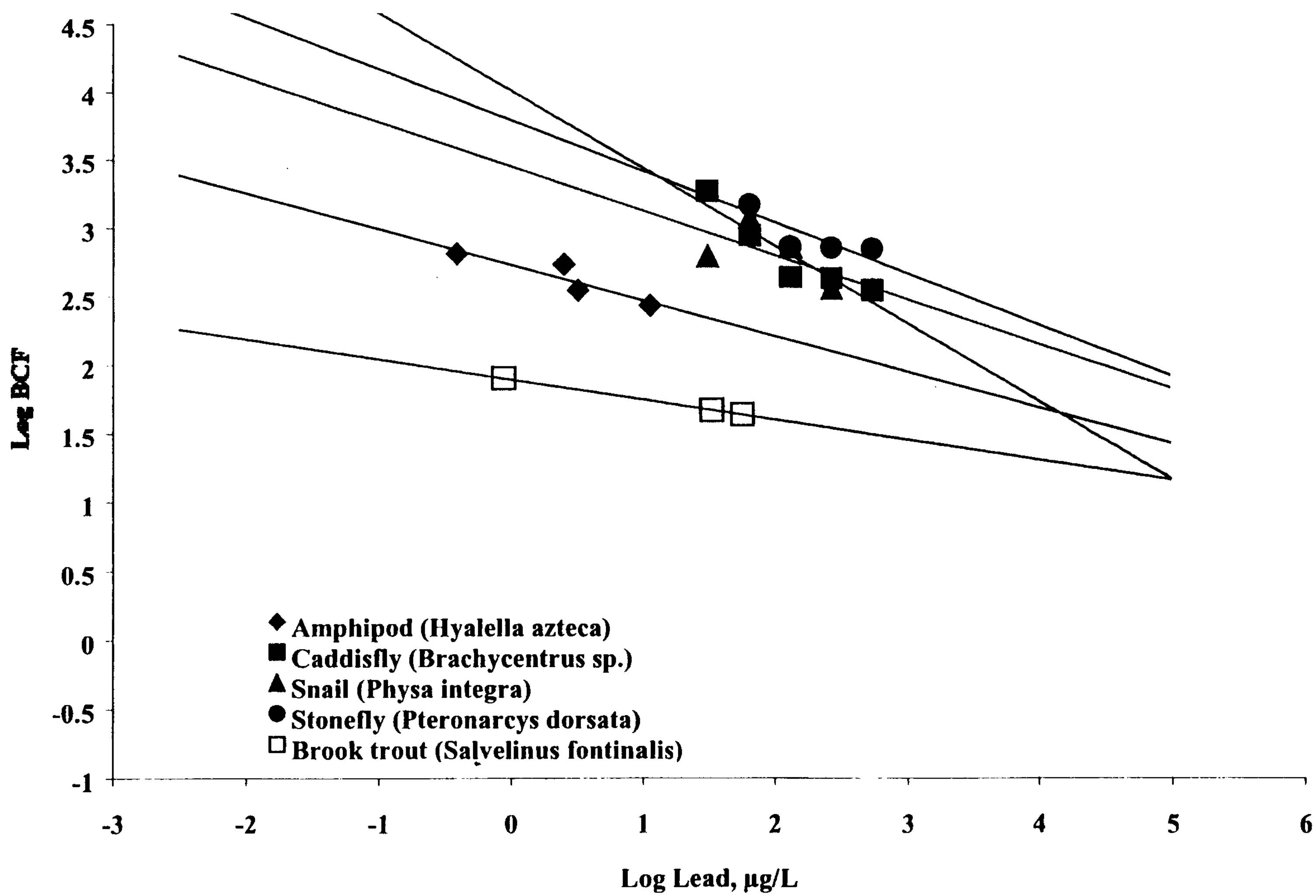


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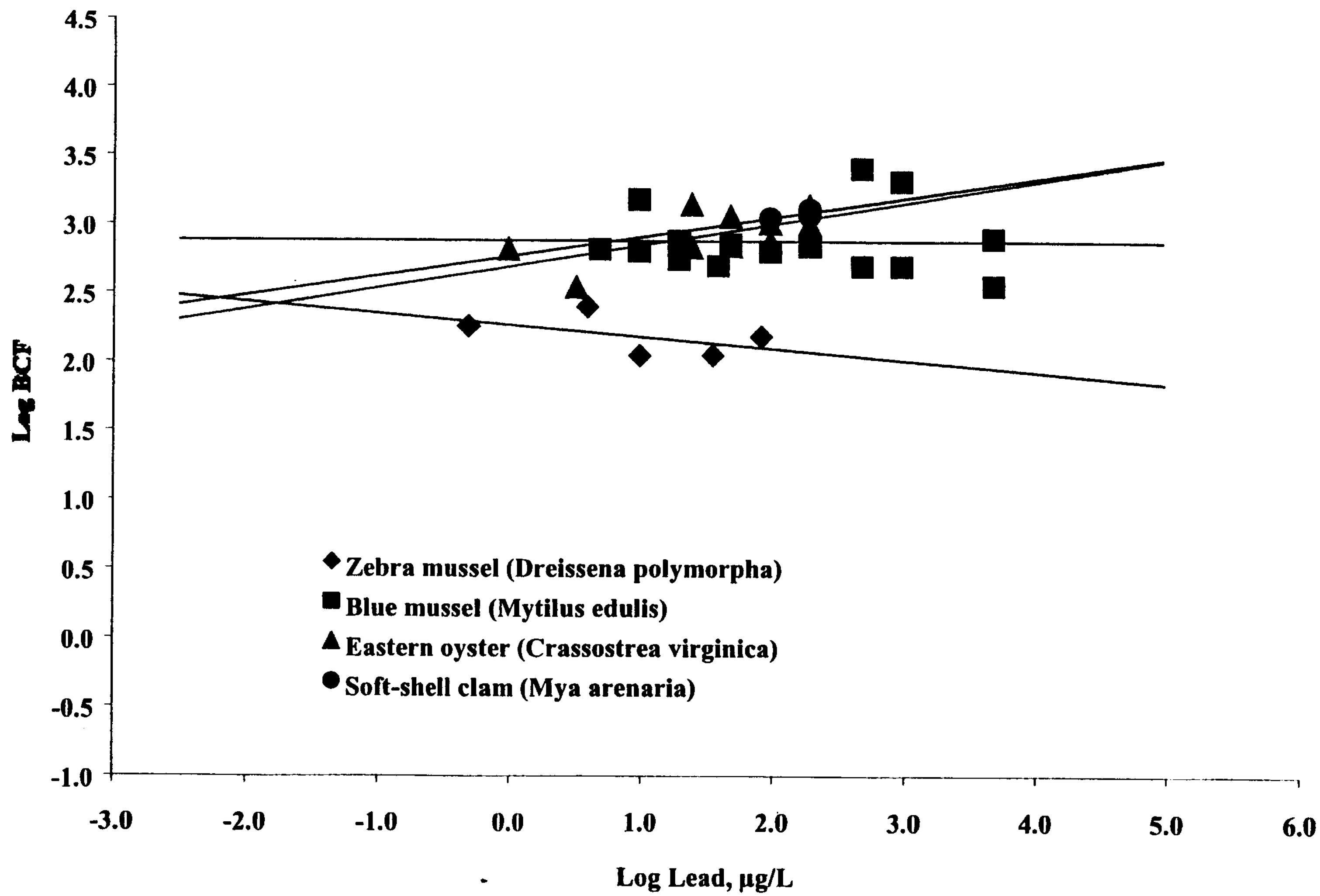
Figure 8. Lead BCFs for Bivalves.



concentrations, *C. edule* still appears to bioaccumulate nickel to a greater extent than other species tested. For example, at 10 µg/L the BCF for *C. edule* is 3,200 compared to 164 for *Mytilus edulis* (Figure 9).

BCFs plotted versus water concentrations again demonstrated an inverse relationship in certain species. Both the bivalves and the fish showed an inverse relationship between BCF and water concentration (Figure 9).

2.3.1.5 Silver

The BCF data for silver are much more limited than for the other five metals. Multiple (i.e., >1) BCFs are available for blue mussel (*Mytilus edulis*) and bluegill sunfish (*Lepomis macrochirus*) (Table 3). An inverse relationship was observed for the blue mussel, while an increasing relationship was observed for bluegill. The BCFs for bluegill sunfish surprisingly increase with increasing exposure concentration (U.S. EPA 1987b). The mechanism for this, if the data are accurate, is unclear. The relationship is only based on two data points from one study, so the results should be interpreted cautiously. Nehring (1976) evaluated silver concentrations in mayflies (*Ephemerella grandis*) and stoneflies (*Pteronarcys californica*) exposed to silver in the laboratory. Silver concentrations were only measured in dead organisms, so the results were not included in the database. However, given that data for silver are lacking, the results from this study are briefly summarized here. Mayfly and stonefly BCFs in dead organisms ranged from 17-84 and 14-37, respectively (assuming a moisture content of 80 percent). As in blue mussels, the BCFs for both insects declined with increasing exposure concentrations.

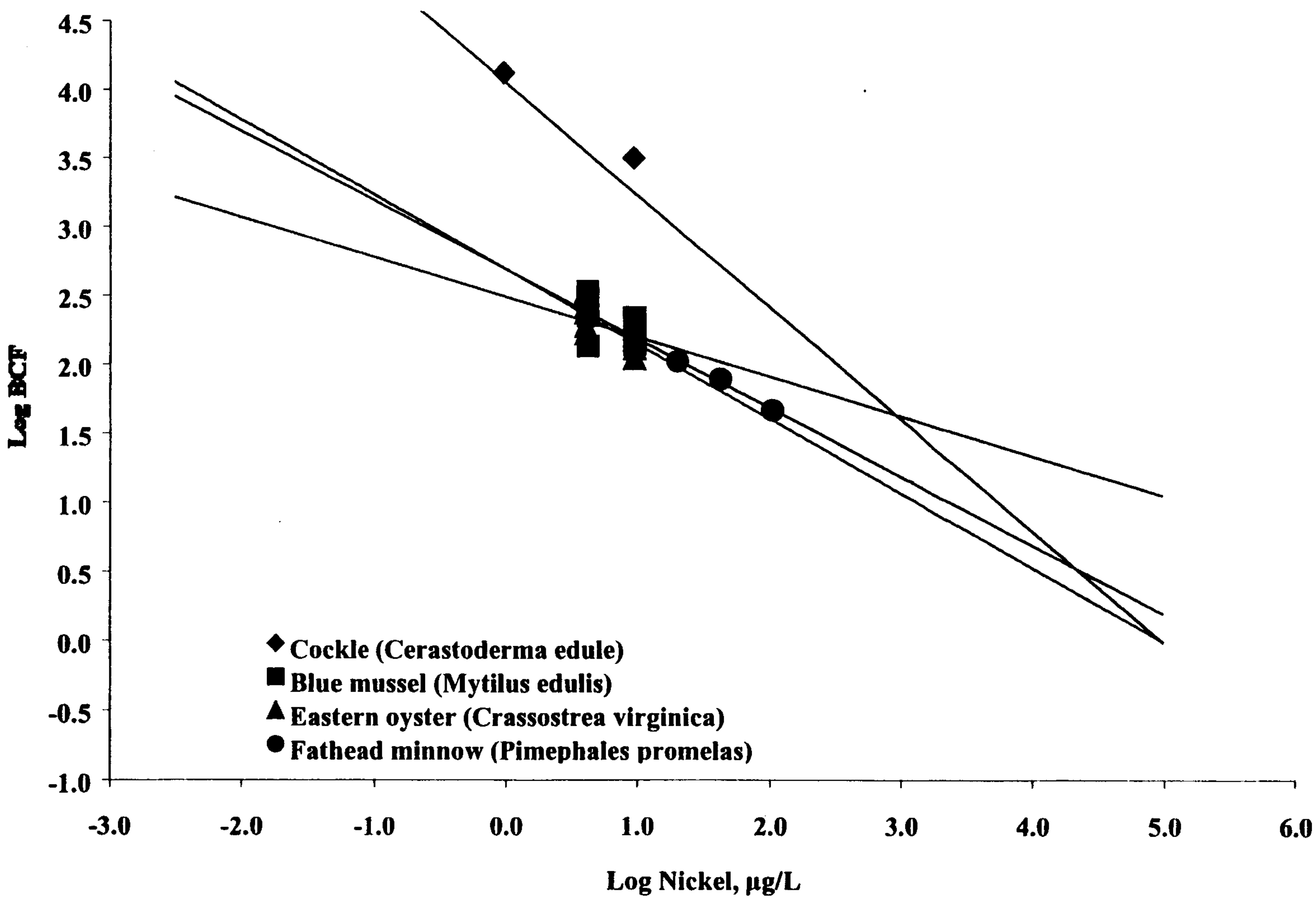
Table 3. Silver BCF data for species with more than one data point.

Species	Water Conc. (µg/L)	Tissue Conc. (µg/kg ww)	BCF
Bluegill (*Lepomis macrochirus*)	10	150	15
	100	15,000	150
Blue mussel (*Mytilus edulis*)	1	765	765
	5	775	155
	10	1,055	106

2.3.1.6 Zinc

Zinc BCFs for algae are quite variable, ranging from 50 in the alga *Olisthodiscus luteus* (Riley and Roth 1971) to 12,000 in the diatom *Thalassiosira pseudonana* (U.S. EPA 1987c). Measured zinc BCFs in non-bivalve invertebrates range from as low as one in the crayfish *Oronectes virilis* (Mirenda 1986a) to 2,640 in the amphipod *Hyalella azteca* (Borgmann et al. 1993). Measured BCFs in some bivalves are higher, being as high as 27,080 in the Eastern oyster (Shuster and Pringle 1969). Measured BCFs in fish are much greater than in the other metals discussed above. In Atlantic salmon (*Salmo salar*), for example, BCFs are as high as 14,000 (Farmer et al. 1979) and as high as 5,800 in the flagfish *Jordanella floridae* (Spehar et al. 1978).

Figure 9. Nickel BCFs for Fish and Bivalves.



As for most of the other metals, an inverse relationship between BCF and water concentration was observed for zinc in bivalves, non-bivalve invertebrates, and fish. The slope was very steep for three of four fish species, but fairly flat for a fourth (*Poecilia reticulata*) (Figure 10). The lack of an observed relationship in *P. reticulata*, however, may simply be due to the narrow concentration range to which fish were exposed (i.e., less than a factor of four difference). This inverse relationship is expected since fish tend to be active regulators of zinc (Phillips and Rainbow 1989). The BCF-water concentration relationship for two species of amphipods are remarkably similar (Figure 11). This strong negative relationship is not surprising since the regulatory strategy of some amphipods (e.g., *Echinogammarus pirloti*) approaches that of active regulation (Rainbow and White 1989).

This amphipod technically does not actively regulate, but its uptake of zinc is so slow that it appears to actively regulate via growth dilution (Rainbow and White 1989). Lastly, inverse relationships were observed in four species of bivalves (Figure 12). It is somewhat surprising that a strong decreasing trend is observed for the Eastern oyster since some oyster species are known to store high zinc concentrations in detoxified granules (George et al 1978). In organisms with this regulatory strategy, the BCF tends to be constant. The apparent decreasing relationship may simply be a function of limited data for the Eastern oyster.

2.3.2 Interpretation

Several observations can be made from the data presented. First, the slopes of the BCF and water concentration relationships for the essential metal zinc are much steeper than those for the non-essential metal lead. Even for non-bivalve invertebrates, the slopes for lead tend to be much more shallow for most species. This provides further support that zinc is more actively regulated than lead. The slopes for lead and zinc appear to represent the two extremes. The slopes for other metals, such as cadmium and copper, tend to fall somewhere between those for lead and zinc. The mechanistic reasons why this occurs is unclear.

Second, these examples provide additional evidence that BCFs are a function of regulatory mechanisms and that single BCFs do not predict the bioaccumulation potential of metals in most organisms due to the inverse relationship between BCF and water concentration. Similar to the algae BCF data plotted in Figure 1, if all BCFs for fish are graphed, a clear decreasing trend is again observed between different species and metals (Figure 13). Again, this demonstrates that metals in fish are regulated by similar mechanisms. Moreover, this figure clearly shows that the inverse relationship between BCFs and water concentrations is important for multiple metals.

The primary principle behind using BCFs in hazard classification is that chemicals with large BCFs have the potential to reach high tissue concentrations and result in long-term direct toxicity or secondary poisoning. The following sections discuss the relationship between BCF, tissue residue concentration, and potential for long-term (chronic) toxicity and/or secondary poisoning.

Figure 10. Zinc BCFs for Fish.

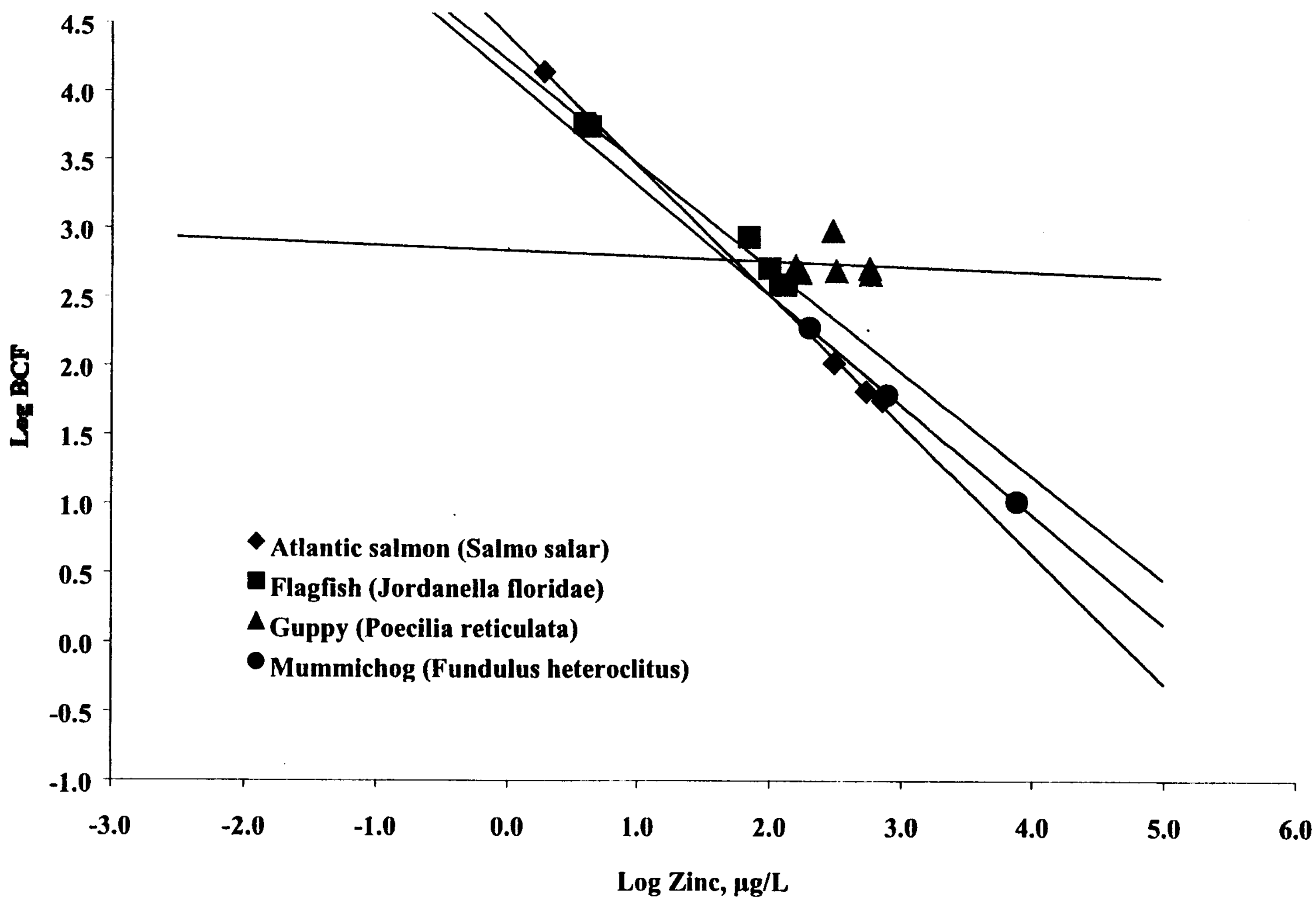


Figure 11. Zinc BCFs for Non-Bivalve Invertebrates.

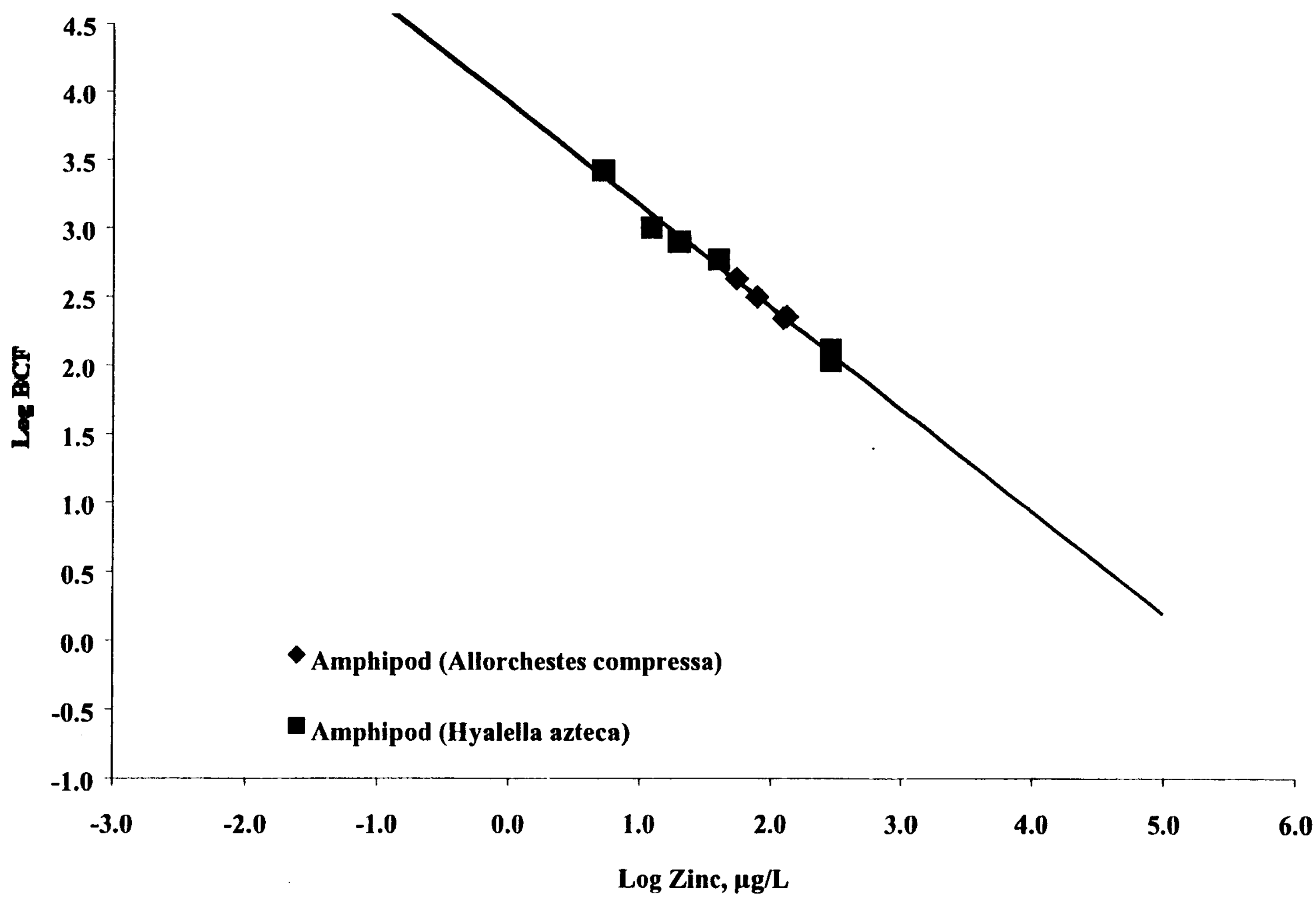


Figure 12. Zinc BCFs for Bivalves.

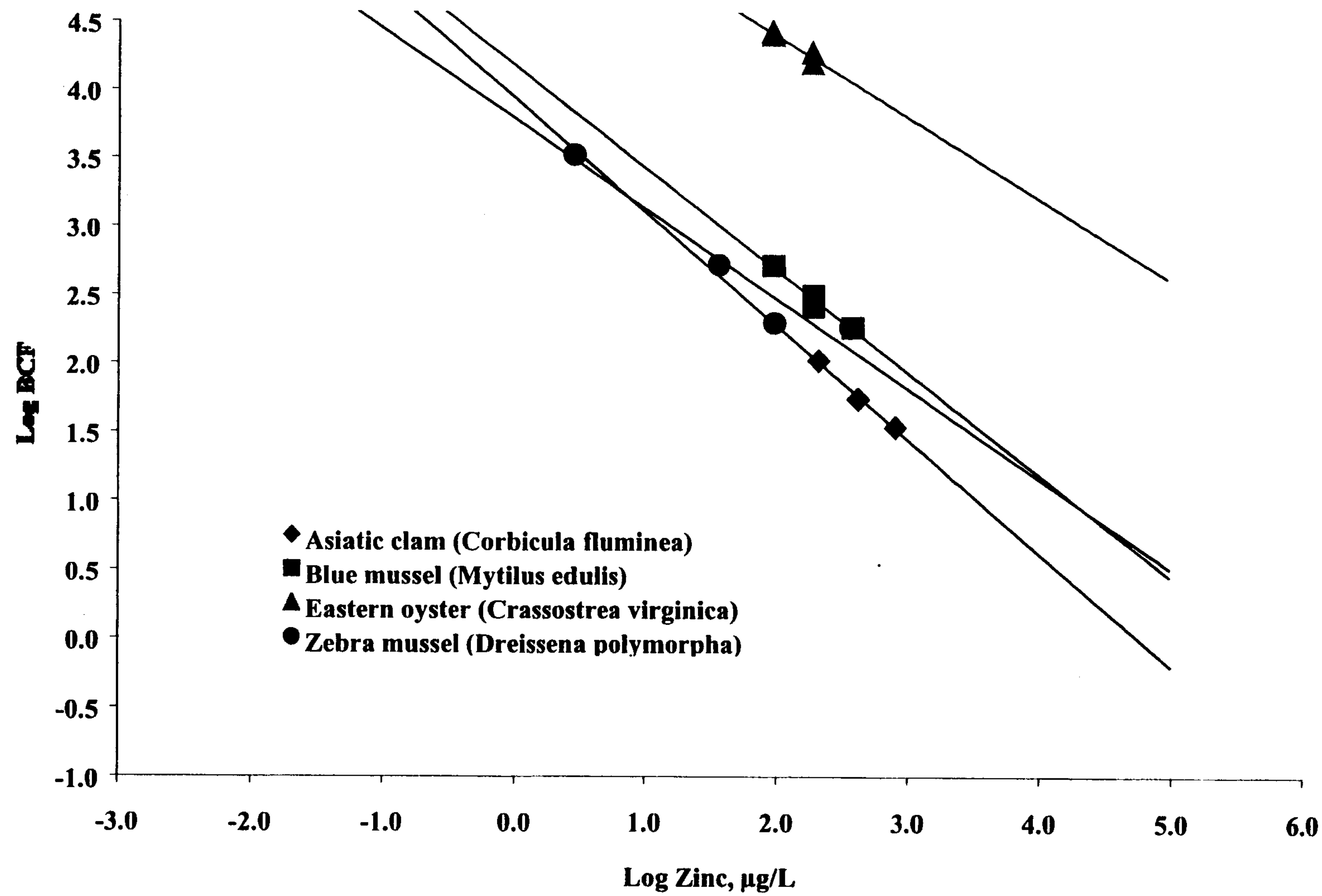
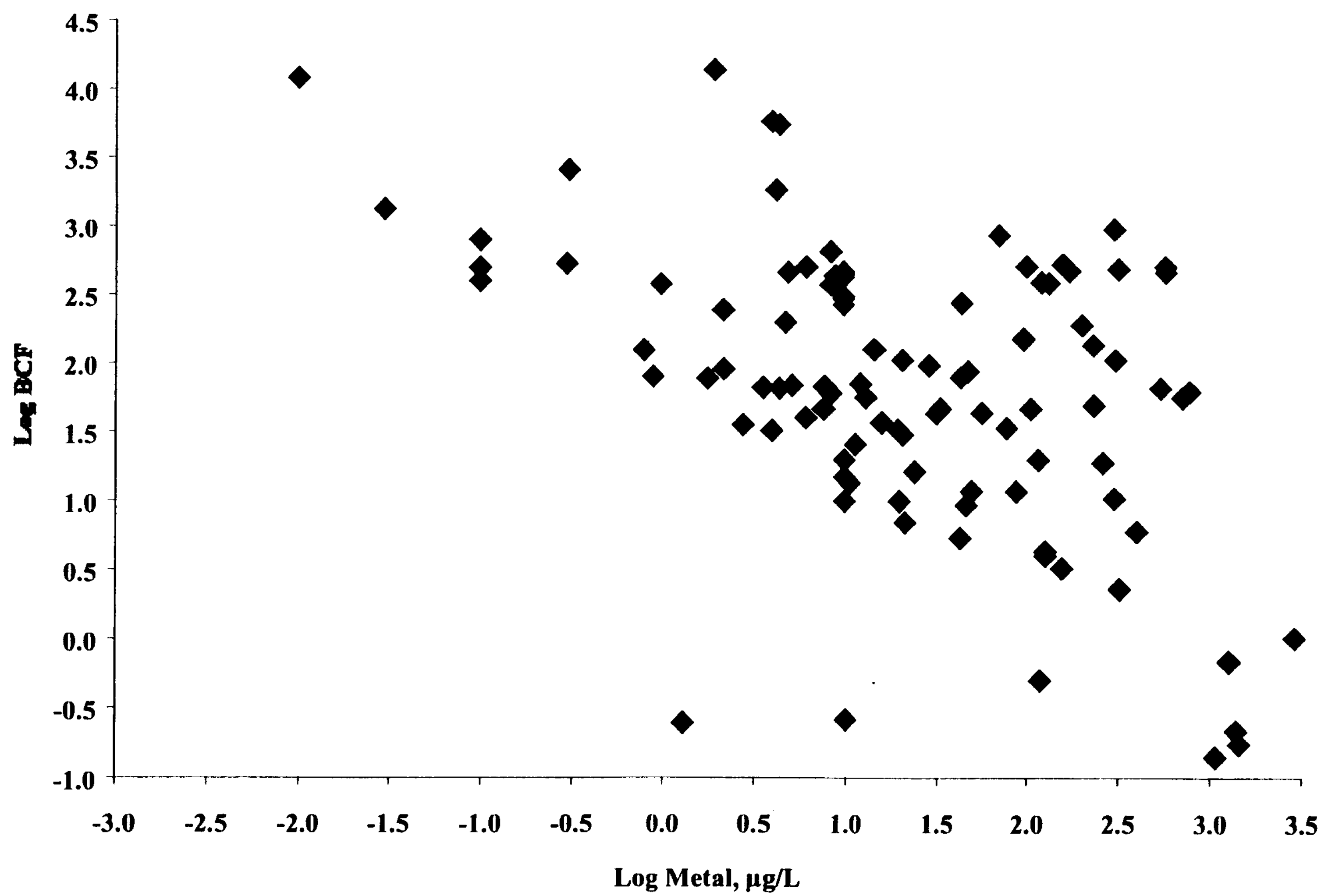


Figure 13. Metal BCFs for Fish.



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2.4 BIOACCUMULATION AS AN INDICATOR OF CHRONIC TOXICITY FOR METALS AND METAL COMPOUNDS

The concept that BCFs can be used as an indicator of chronic toxicity stems from the assumption that larger BCFs are indicative of higher tissue concentrations, which in turn result in direct or secondary toxicity. This concept is primarily relevant to organic chemicals with narcosis as the mode of toxic action (Veith et al. 1985, McCarty 1986). An organic chemical's bioaccumulation potential is often related to its lipophilicity, as measured by its octanol-water partition coefficient (K_{ow}) (Veith and Kosian 1982). Chemicals with a high K_{ow} have a slow uptake rate in organisms and an even slower excretion rate (Veith et al. 1979, Spacie and Hamelink 1982).

This relationship results in two general trends: (1) the acute toxicity of high K_{ow} compounds is relatively low because uptake is limited during short-term exposures; and (2) the low excretion rates can result in chronic toxicity at levels much lower than that observed for acute toxicity (i.e., large acute-chronic ratios). Consequently, organic chemicals with a high K_{ow} have the potential for chronic toxicity at concentrations much lower than those observed for acute toxicity (i.e., the chronic toxicity potential of high K_{ow} compounds is greater). Hydrophobicity, as measured by K_{ow}, can be used to explain and predict toxicant kinetics and effects, but as noted by McCarty (1986), this interrelationship may be of utility for many organics but will not apply to all chemicals. For metals, Winner (1984) states that studies have shown that accumulated metal may be poorly, or even negatively, correlated with toxicity. For example, a negative correlation between whole body bioaccumulation and toxicity of copper in rainbow trout has been demonstrated (Dixon and Sprague 1981), possibly due to the induction of metallothionein (Winner 1984) and due to a lack of measurement of copper at the site of action where the toxicity occurs.

Simply put, the bioaccumulation potential of a metal based on whole body measurements is not indicative of its toxicity because aquatic organisms have regulatory mechanisms for actively excreting excess metal and/or for storing excess metal in detoxified forms. Additionally, metals only appear to become toxic when these regulatory mechanisms are overwhelmed (Phillips and Rainbow 1989, Bryan 1979) and when the concentration at a site of toxic action exceeds a toxic threshold. Obviously at high enough exposures, bioaccumulation in an organism overloads regulatory mechanisms and exceeds a toxicity threshold at a site of toxic action. This unpredictability of BCFs stems from two primary factors. First, as demonstrated above, BCFs are a poor indicator of bioaccumulation potential in a variety of organisms because BCFs are often inversely related to exposure concentration. The data on rainbow trout exposed to cadmium in the previous section provided a good example of this. Second, bioaccumulation potential is a poor indicator of toxicity because many organisms can store metals in detoxified forms. Consequently, the bioaccumulation potential in these organisms may be high, but the potential for toxicity is negligible. Issues associated with the relationship between bioaccumulation and toxicity of metals are discussed further below with specific examples provided.

2.4.1 Relationships Between BCFs and Chronic Toxicity

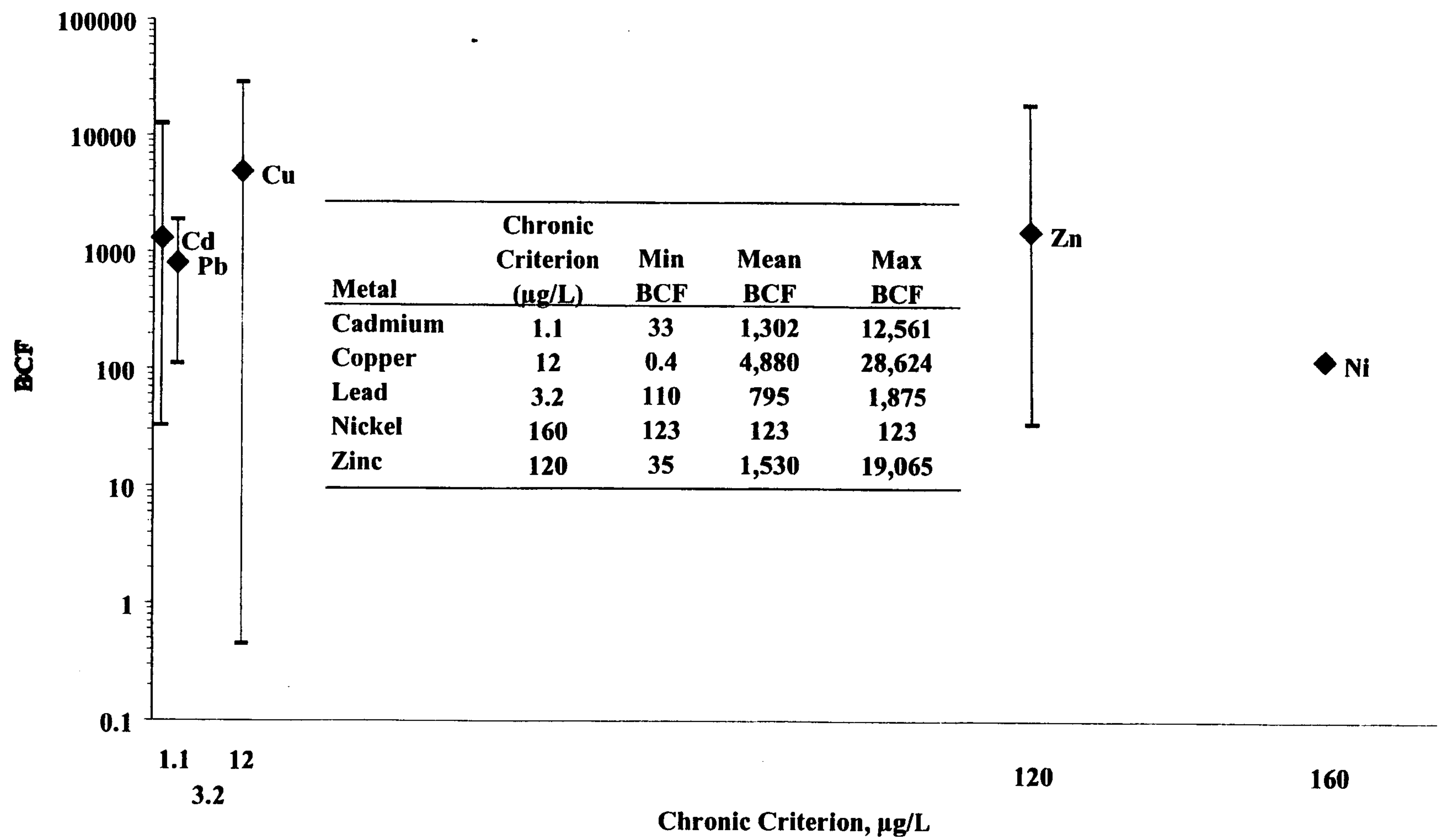
To demonstrate the lack of relationship between the magnitude of the whole body BCF and the chronic toxicity of metals, a series of figures were developed to graphically display the data. In Figure 14, the mean, minimum, and maximum BCFs (fish and invertebrates) for five metals are plotted against their chronic toxicity as defined by the U.S. EPA ambient water quality criteria for each metal (U.S. EPA 1985b,d,e, U.S. EPA 1986, U.S. EPA 1987c). Other regulatory agencies have developed alternative criteria, but the general relationships between metals are the same. As shown in the plot, cadmium is the most toxic of the metals evaluated and nickel the least toxic. Because BCFs for metals are highly dependent on exposure concentration for many aquatic organisms, we only plotted the BCFs that were based on exposure concentrations at the chronic criterion and ten times the chronic criterion. For example, the U.S. EPA chronic criterion for cadmium is 1.1 µg/L; therefore, only BCFs derived at water concentrations between approximately 1 and 11 µg/L were plotted. This ensures that very high or very low BCFs for a metal, based on exposure to very low or very high metal concentrations in water, are not over represented in the figures and adding bias to the comparison between BCFs and chronic toxicity potential. As shown in Figure 14, no relationship between the magnitude of the BCF and chronic toxicity is apparent based on this figure. For cadmium and zinc, the mean, minimum, and maximum BCFs are almost identical despite the chronic criterion for cadmium being 120 times lower than the zinc criterion.

Evaluating a single species and a single metal, the study by Borgmann et al. (1978) provides further evidence that there is no relationship between the magnitude of a BCF and direct toxicity. Borgmann et al. exposed aquatic snails (*Lymnaea palustris*) to lead nitrate for 120 days. Results from this study are summarized in Table 4. The study demonstrated that increased toxicity is not necessarily observed in organisms with larger BCFs. The percent survival in snails with a lead BCF of 2,500 was not less than the percent survival in snails with a lead BCF of 304. Further, the same levels of toxicity are not observed in organisms with similar BCFs. The percent survival in three groups of snails, all with lead BCFs of approximately 2,500, ranged from 15-79 percent. At a low water concentration, the lack of a relationship between the BCF and toxicity is again a function of the inverse relationship between the BCF and water concentration. At higher water concentrations the BCF is fairly constant, demonstrating that tissue concentrations were increasing in proportion to exposure concentrations. At these concentrations, the results demonstrate that toxicity is not related to the BCF but, rather, must be related to the actual tissue concentration.

Table 4. Relationship between lead BCFs and toxicity in the snail *Lymnaea palustris* (Borgmann et al. 1978).

Water Conc. (µg/L)	Whole Body Tissue Conc. (mg/kg ww)	BCF	% Survival
3.8 (control)	1.2	300	69
12	30.4	2,500	79
19	47.0	2,500	39
31	71.2	2,300	15
54	NR	NA	2
97	NR	NA	0

Figure 14. Relationship Between BCFs and Exposure Concentrations at the U.S. EPA Chronic WQC and 10 times the WQC (mean, minimum, and maximum).



Metal	Chronic Criterion (µg/L)	Min BCF	Mean BCF	Max BCF
Cadmium	1.1	33	1,302	12,561
Copper	12	0.4	4,880	28,624
Lead	3.2	110	795	1,875
Nickel	160	123	123	123
Zinc	120	35	1,530	19,065

In organisms that do not regulate metals by active excretion, but store them in detoxified forms, there also does not appear to be any relationship between bioaccumulation potential and toxicity. Since these aquatic organisms can bioaccumulate large amounts of metal without overloading their storage mechanisms, bioaccumulation potential is clearly not indicative of the metal's potential to exert toxicity. The barnacle *Elminius modestus*, for example, accumulates zinc at a high rate with no apparent significant excretion (Rainbow 1996). Concentrations in some barnacles may reach 100,000 mg/kg dw without any known deleterious effect on the organism.

Based on the BCFs for multiple metals (arsenic, cadmium, copper, lead, nickel, selenium, and zinc) presented in previous figures, it is clear that the BCF for most types of organisms is dependent on the exposure concentration. Although the observed BCFs are clearly variable and overlap between metals, no relationship exists between the magnitude of a metal's BCF and its direct toxicity potential.

2.4.2 <u>Species Sensitivity Distributions</u>

As explained above, metal BCFs and bioaccumulation potential are influenced by the regulatory mechanisms used by aquatic organisms. The toxicity of metals to aquatic biota is also directly related to their regulatory mechanisms because toxicity is observed when these regulatory mechanisms are overloaded (Phillips and Rainbow 1989). Accordingly, the regulatory mechanism influences metal bioaccumulation, BCFs, and toxicity. To further evaluate the relationships between regulatory mechanism, bioaccumulation potential, and toxicity, large databases of acute toxicity data for metals were analyzed. It was not possible to analyze chronic toxicity data because they are available for only a limited number of species. These databases contain toxicity data for diverse groups of freshwater and marine organisms with different regulatory strategies, including: cladocerans, copepods, amphipods, bivalves, oligochaete worms, polychaete worms, gastropods, decapods, aquatic insects, and several fish species. In evaluating the species sensitivity distributions for cadmium, copper, nickel, silver, and zinc, it is clear that certain taxonomic groups have similar relative sensitivities to these metals (Brix et al. 2000a,b).

As discussed in Section 3.2, the marine decapod *P. elegans* regulates copper and zinc over a wide range of metal concentrations. Other decapods, including lobsters and crabs, are also known to actively regulate copper and zinc. Apparently, active regulation of essential metals by decapods is not necessarily restricted to older life stages of organisms, such as juveniles and adults. Larvae of the prawn *Palaemon serratus*, for example, have been reported to regulate zinc (Devineau and Amiard-Triquet 1985). It is presumed that embryos lack sufficiently developed organs to actively regulate or detoxify metals. As such, these life stages are assumed to be more sensitive to metals. The marine amphipod *E. pirloti* accumulates zinc, but the rate of net uptake is very slow. In addition, this amphipod also accumulates copper with no evidence of active regulation. At the other extreme, the barnacle *E. modestus* accumulates high levels of copper and zinc in detoxified granules with no significant excretion. The data for marine gastropods suggest that regulation of zinc is species-specific. For example, *Littorina littorea* is known to store zinc and other metals in detoxified granules (Mason and Nott 1981), while Kaland et al. (1993) suggest that *Nassarius reticulatus* may be able to actively regulate zinc to some degree. As for non-essential metals, the decapod, amphipod, and barnacle all accumulate cadmium with no active regulation. Given the above information, the relative sensitivities of these organisms, and related organisms, were then compared to determine whether a relationship may exist between sensitivity and regulatory mechanism.

2.4.2.1 Cadmium

Saltwater Organisms. The available toxicity data for cadmium suggest that, overall, decapod crustaceans are the most sensitive taxonomic group (Figure 15). Toxicity data are available for multiple life stages of crabs, shrimp, and lobsters, and early life stages are more sensitive than older life stages. This is not surprising given that early life stages are expected to be more sensitive due to their underdeveloped regulatory abilities and that older life stages of decapods tend to be active regulators that are unable to detoxify excess metal. Amphipods, in contrast to decapods, do not appear to excrete excess metal or have well developed storage mechanisms for metals, rather, they tend to regulate metals by having a very slow uptake rate. As a result, amphipods would be expected to have a moderate to high sensitivity to cadmium after a sufficient exposure time. However, amphipods actually appear to be of moderate to low sensitivity. The basis for this result is not entirely clear, but it may be that their relatively slow uptake rates reduce their relative sensitivity in short-term acute toxicity tests. At least some species of gastropods are known to bind metals with metallothionein and other organoproteins (Kaland et al. 1993). This may explain why gastropods tend to have a relatively low sensitivity to cadmium (Figure 15). Worms appear to have variable regulatory strategies between species, so it is not surprising that the sensitivities of worms relative to other organisms is quite variable (Figure 15). Overall, the data suggest that organisms with negligible regulatory abilities (e.g., embryos) or active regulation tend to be more sensitive to cadmium. These organisms tend to have limited mechanisms for storing detoxified metals and, hence, cannot tolerate very high levels of a non-essential metal such as cadmium.

Freshwater Organisms. The freshwater taxa most sensitive to cadmium are cladocerans and amphipods, while the least sensitive taxa include insects and worms (Figure 16). Limited data were identified in the literature on the mechanisms used by cladocerans to regulate metals, but since they do not bioaccumulate cadmium to high levels (Figure 3), it is unlikely that they are able to store large amounts of detoxified cadmium. Griffiths (1980) observed calcium granules in *Daphnia magna* exposed to cadmium, but it is not clear if these granules act as a detoxifying mechanism. Bodar et al. (1990) and Stuhlbacher et al. (1992) studied cadmium resistance in *Daphnia magna* and determined that resistance was a physiological response, not hereditary. *Daphnia* that were pre-exposed to cadmium tended to accumulate more cadmium than daphnids that were not pre-exposed. It is possible that cadmium was being bound by metallothionein-like proteins (Bodar et al. 1990, Stuhlbacher et al. 1992). Given that there is little evidence to suggest that any organism can actively regulate the non-essential metal cadmium, it is likely that the limited storage capacity that daphnids may have when not pre-exposed to cadmium is quickly overwhelmed and toxicity results at relatively low concentrations.

Amphipods, in general, also do not appear to have well developed storage mechanisms for metals. Rather, they tend to regulate metals by having a very low net uptake rate. Similar to cladocerans, therefore, they may be among the more sensitive species to cadmium because they have a limited ability to store the metal in a detoxified form. In contrast, aquatic insect larvae do store metal. Consequently, these organisms generally have the capacity to store large amounts of metal in a

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Figure 15. Relative Acute Sensitivities of Saltwater Organisms to Cadmium (from Brix et al. 2000b).

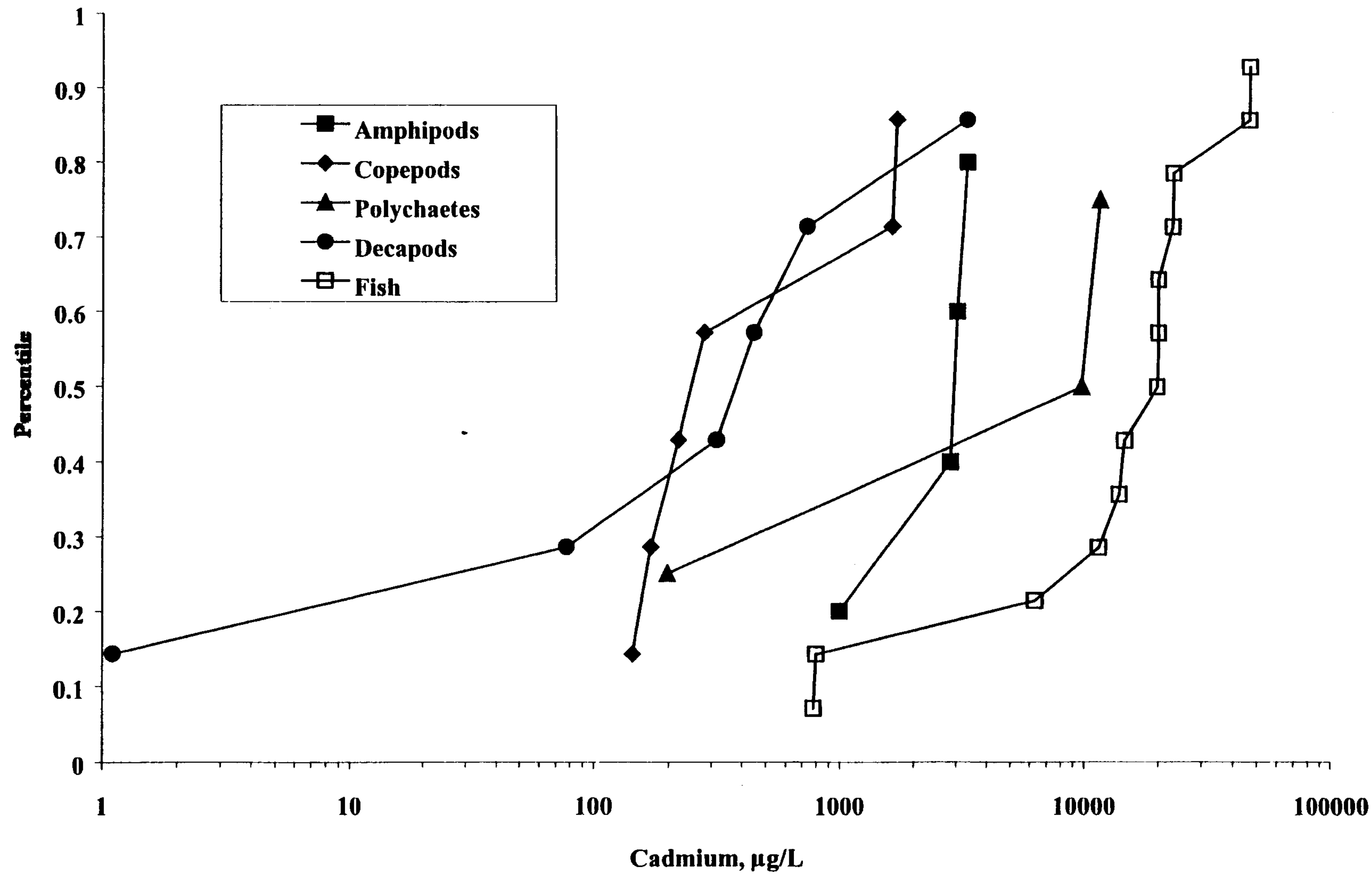
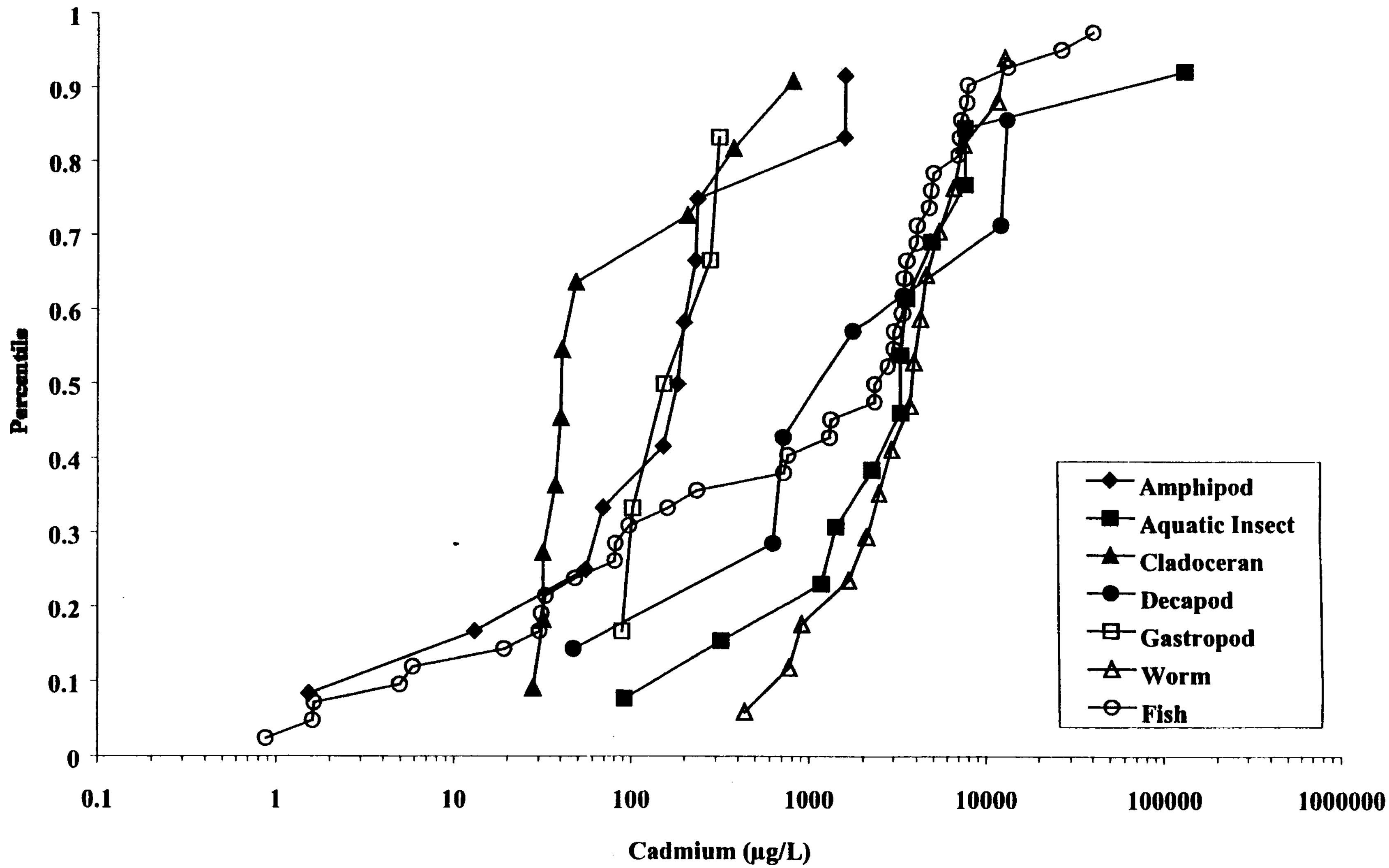


Figure 16. Relative Acute Sensitivities of Freshwater Organisms to Cadmium (from Brix et al. 2000b).



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detoxified form. There are many factors that influence the sensitivities of organisms to a metal, but the above examples suggest that the regulatory mechanisms of aquatic biota may be a significant factor. Since the regulatory strategy of an aquatic organism profoundly influences the bioaccumulation potential of a metal within an organism, these examples provide further evidence that bioaccumulation potential cannot be correlated with the potential for toxicity. The above examples, may even suggest an opposite relationship exists, i.e., bioaccumulation potential and toxicity potential of metals are inversely related as a function of metal regulatory strategy. Organisms that bioaccumulate large concentrations of metals do so because they have the necessary storage mechanisms to detoxify the metal; organisms without the ability to store metals in detoxified forms generally are more sensitive and bioaccumulate lower concentrations of metals.

2.4.2.2 Copper

Saltwater Organisms. The most sensitive species to copper that have been tested include early life stages of bivalves and fish (Figure 17). Of the five most sensitive species to copper, the life stage for four was the embryo (three bivalves and one fish species) and newly hatched nauplii in the fifth (a copepod). As discussed above, early life stages of organisms such as these embryos, in particular, generally have underdeveloped regulatory abilities compared to adult organisms. Toxicity data are also available for larvae of the barnacle (*Balanus improvisus*). Presumably due to the lack of detoxification mechanisms at this life stage, larvae of this barnacle are of moderate sensitivity, rather than low sensitivity as would be expected given the high copper storage capacity of adults. Toxicity data are available for five species of decapods. Given that decapods can generally actively regulate copper, it was expected that these organisms would be among the more sensitive invertebrates. Instead, however, the decapods have a wide range of sensitivities to copper. Dungeness crab (*Cancer magister*) larvae are fairly sensitive, but green crab larvae (*Carcinus maenas*) are over an order of magnitude less sensitive. The polychaete *Nereis* does not appear to regulate copper (Leland and Kuwabara 1985), but stores it in detoxified forms. This is consistent with the available toxicity data for copper, as *Nereis diversicolor* is one of the least sensitive marine species that have been tested.

Freshwater Organisms. The species sensitivity distribution for copper is very similar to that observed for cadmium (Figure 18). Cladocerans and amphipods again appear to be the most sensitive invertebrates, while insects appear to be the least sensitive. Although copper is an essential metal and cadmium is non-essential, it is probably due to similar mechanisms that this same relative pattern between taxa is observed. One difference being that organisms can tolerate higher concentrations of copper than cadmium.

2.4.2.3 Nickel

Saltwater Organisms. Given that the toxicity data for nickel are much more limited than for cadmium and copper, and the strategies used by aquatic organisms to regulate nickel are not well known, it is difficult to identify relationships between sensitivity and regulatory strategy. The available data demonstrate that mysids are among the most sensitive organisms that have been tested (Figure 19). The data for bivalves again demonstrate the importance of life stage. The two most sensitive bivalve data points are based on the embryo life stage, while the least sensitive life

Figure 17. Relative Acute Sensitivities of Saltwater Organisms to Copper (from Brix et al. 2000b).

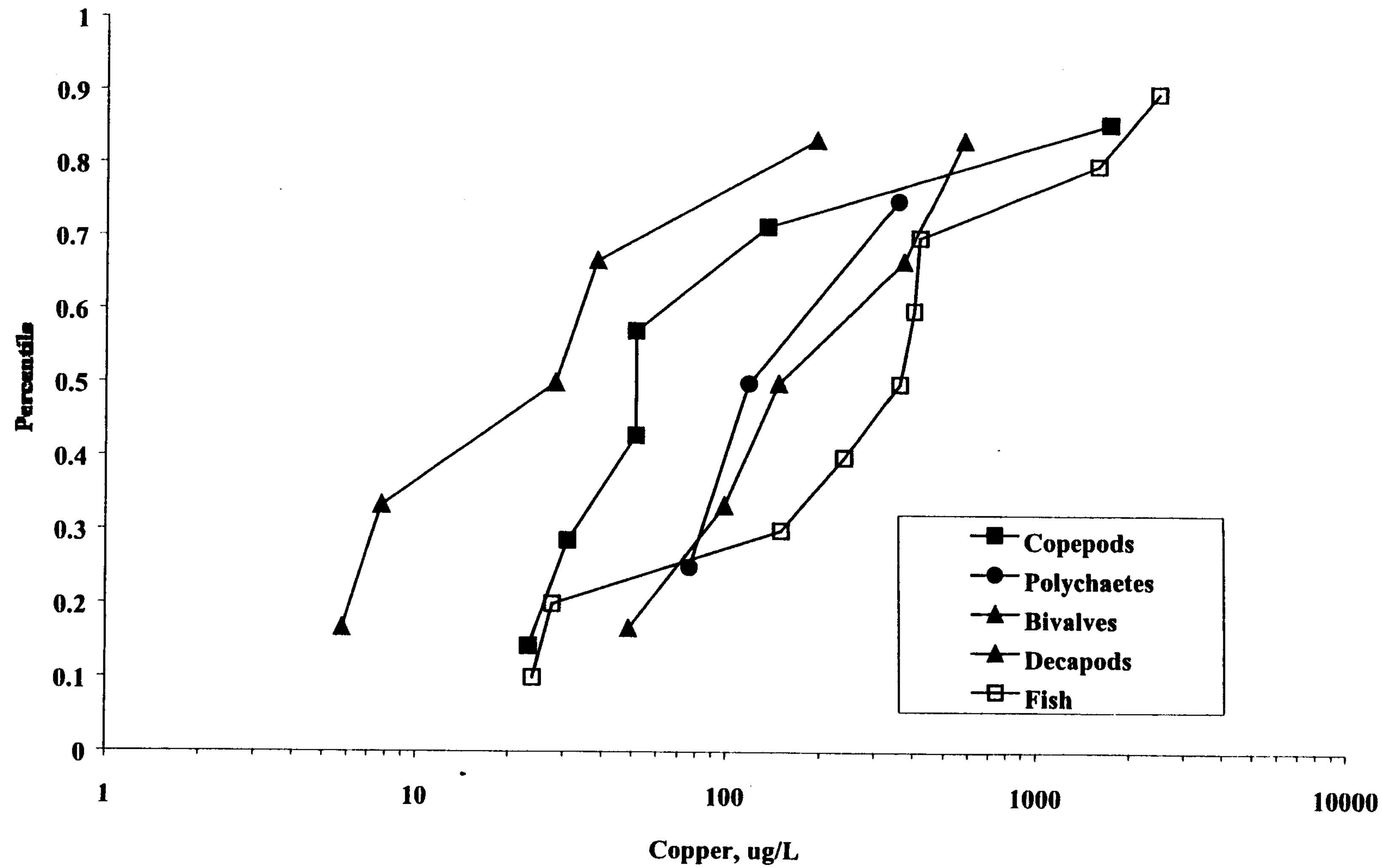

Figure 18. Relative Acute Sensitivities of Freshwater Organisms to Copper (from Brix et al. 2000a).

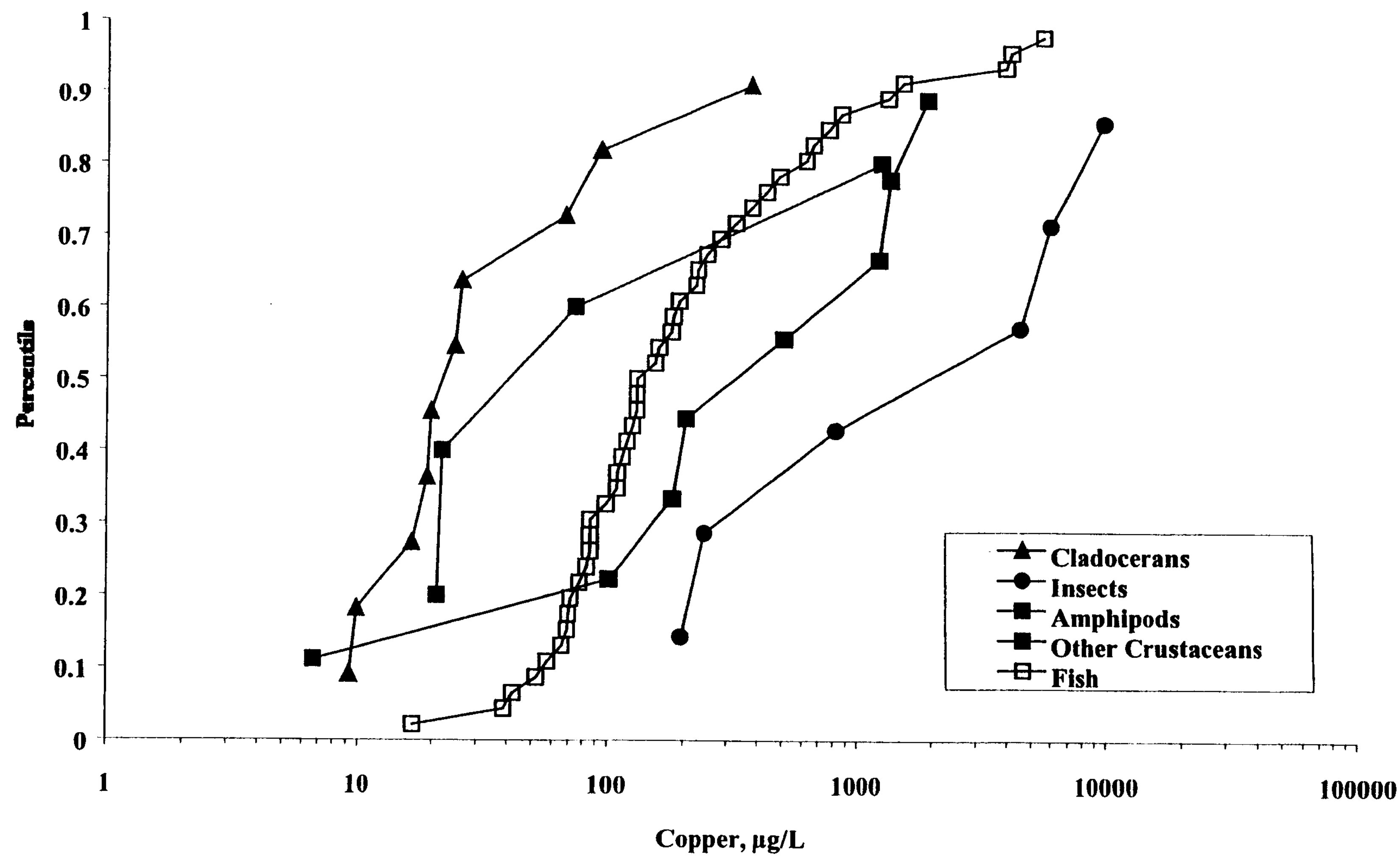

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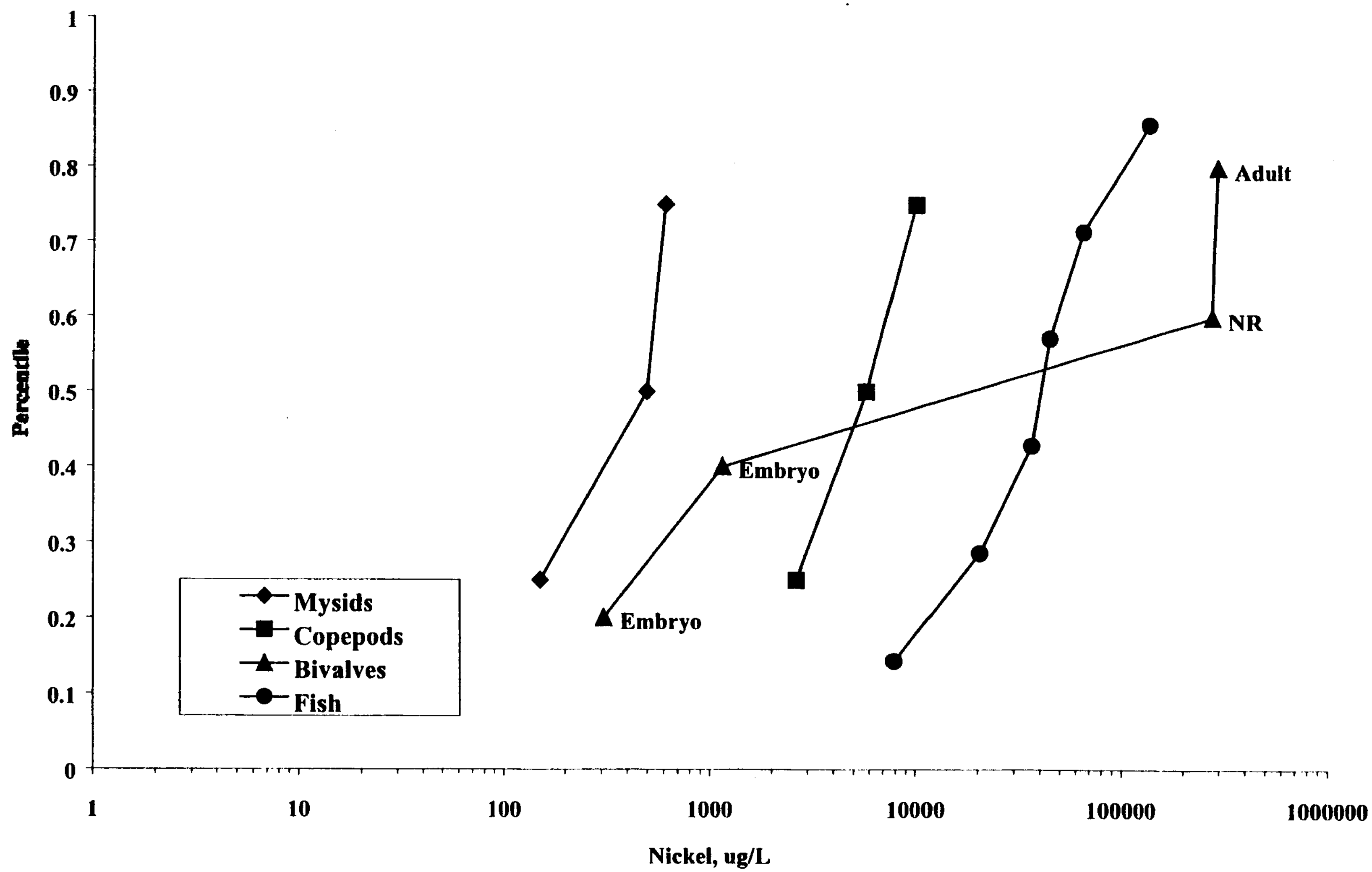
Figure 19. Relative Acute Sensitivities of Saltwater Organisms to Nickel (from Brix et al. 2000b).



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stage is based on an adult (the life stage of the third most bivalve was not reported). Bivalve embryos do not have the ability to actively regulate or sequester excess nickel and, therefore, are much more sensitive. Older life stages are much less sensitive despite the fact that they are likely to have much higher BCFs.

Freshwater Organisms. The nickel toxicity data for freshwater organisms again demonstrate that cladocerans are among the most sensitive species and aquatic insects are among the least sensitive species (Figure 20). Although the regulatory mechanisms have not been well studied, it can probably be assumed that cladocerans have limited ability to sequester excess nickel, while insects can sequester large amounts.

2.4.2.4 Silver

Saltwater Organisms. Toxicity data for silver are not available for a sufficient number of saltwater species to evaluate the relationship between sensitivity and regulatory strategy.

Freshwater Organisms. Silver toxicity data for freshwater organisms are somewhat limited, but again cladocerans, along with amphipods, appear to be the most sensitive organisms tested (Figure 21). Insects are again generally less sensitive, suggesting that they can sequester silver in non-toxic forms.

2.4.2.5 Zinc

Saltwater Organisms. Like copper, the most sensitive species to zinc are bivalve and fish embryo-larvae (Figure 22). As demonstrated in the figure, storage mechanisms develop with age and the sensitivities of juvenile and adult bivalves lessen compared to embryo-larval life stages. This is an important point because it provides further evidence that BCFs are not related to the sensitivity of organisms to metals. Relatively high BCFs are often associated with adult bivalves, but adult bivalves tend to be some of the least sensitive organisms/life stages to metal toxicity. Interestingly, decapods have a range of sensitivities, even considering that many are known to be active regulators. This again may be explained by the slow uptake rate for some species.

Freshwater Organisms. The relative sensitivities of various taxa to zinc is again similar to the relative sensitivities to copper and cadmium (Figure 23). Cladocerans and insects are again the most and least sensitive taxonomic groups tested, respectively. The toxicity data for zinc further support that the sensitivity of organisms is a function of their regulatory mechanisms, and not the bioaccumulation potential of a metal.

2.4.2.6 Conclusions from Species Sensitivity Distributions

Among freshwater organisms, cladocerans tend to be the most sensitive taxonomic group to all of the metals evaluated, while insects tend to be the least sensitive, or among the least sensitive, taxonomic group (Brix et al. 2000a,b). The difference in sensitivities between these two taxonomic groups can probably be partially explained by their different regulatory strategies. As discussed, no data were identified on whether cladocerans actively regulate essential metal. Their ability to

Figure 20. Relative Acute Sensitivities of Freshwater Organisms to Nickel (from Brix et al. 2000b).

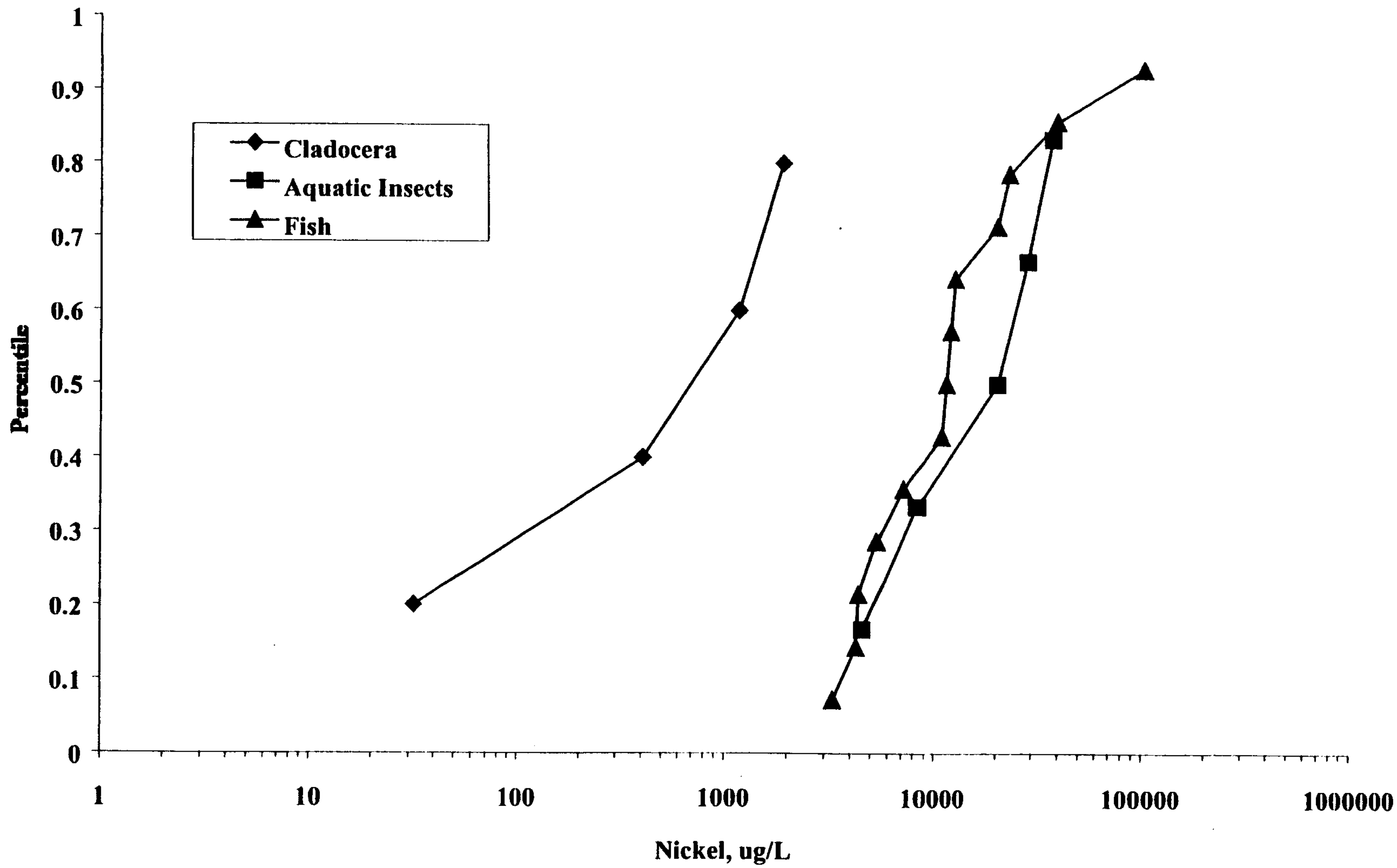

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Figure 21. Relative Acute Sensitivities of Freshwater Organisms to Silver (from Brix et al. 2000b).

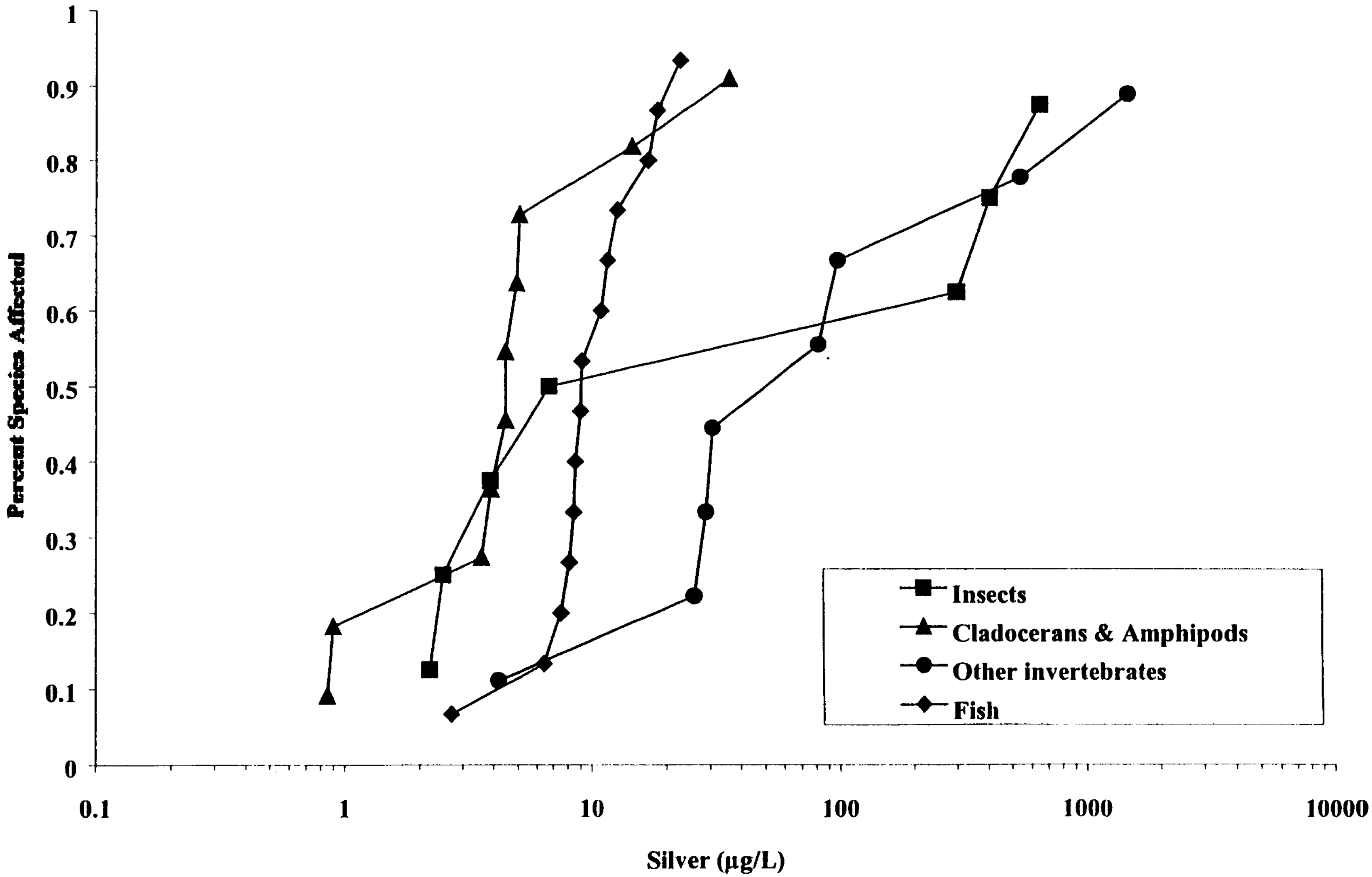

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Figure 22. Relative Acute Sensitivities of Saltwater Organisms to Zinc (from Brix et al. 2000b).

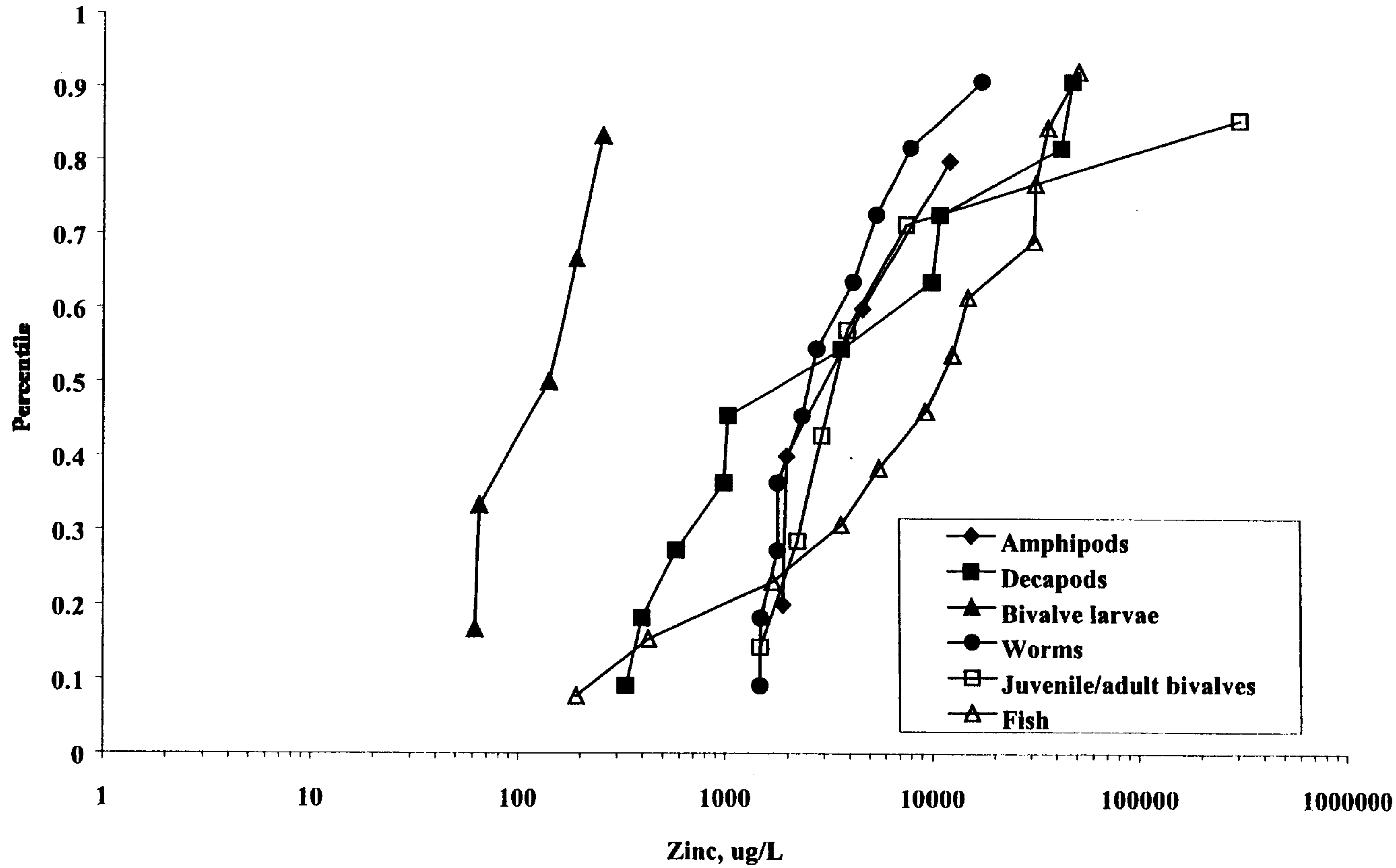
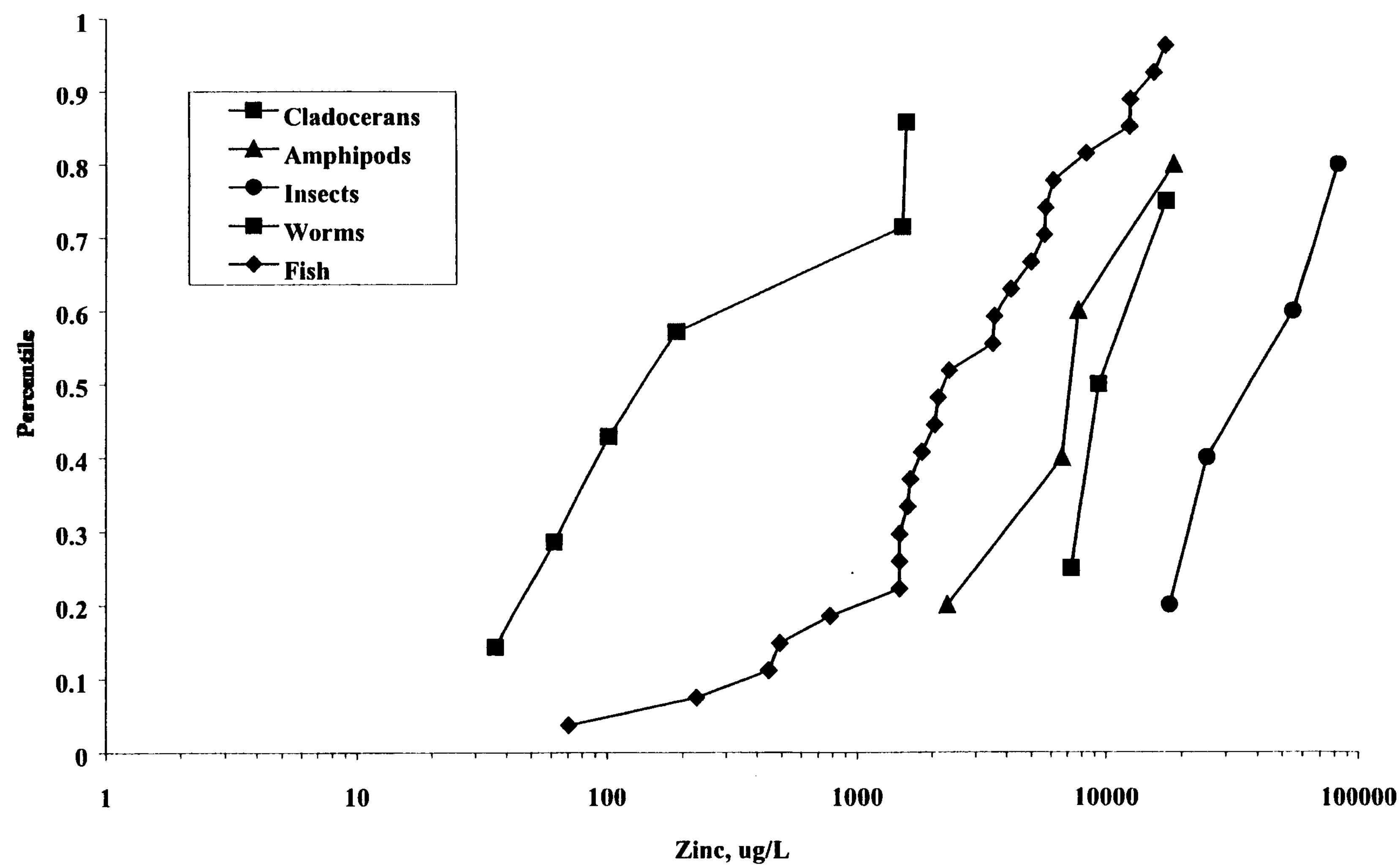


Figure 23. Relative Acute Sensitivities of Freshwater Organisms to Zinc (from Brix et al. 2000b).



detoxify metals using metallothionein-like protein, however, seems to be limited unless they are pre-exposed to a metal. Because they have limited ability to detoxify metals, their regulatory mechanisms are rapidly overwhelmed and toxicity results. Conversely, aquatic insect larvae appear to be net accumulators of heavy metals (Rainbow and Dallinger 1993). Because insect larvae can store metals in non-toxic forms, they can regulate larger metals concentrations and, accordingly, are less sensitive to the toxic effects of metals.

A similar pattern is observed for saltwater organisms. The most sensitive species tend to be embryos/larvae with undeveloped or poorly developed regulatory systems or certain decapods that are known to be active regulators. These types of organisms have a limited ability to sequester excess metals and, as a result, tend to be among the more sensitive species. Other organisms, such as polychaetes tend to be less sensitive – many of these organisms are known to store metal in detoxified forms. Consequently, they have a greater ability to sequester excess metal and are not as sensitive.

These relationships provide further support that bioaccumulation potential has no relationship to toxicity. Several cladoceran and aquatic insect BCFs are available for cadmium, for example (Figure 24). As the figure shows, most BCFs for aquatic insects are greater than those for cladocerans, despite the sensitivities of cladocerans being much greater than for insects. If anything, an inverse relationship between bioaccumulation potential and sensitivity may exist since those organisms that can bioaccumulate metals in non-toxic forms tend to be the least sensitive.

2.5 SECONDARY POISONING AND BIOMAGNIFICATION OF METALS AND METAL COMPOUNDS

Secondary poisoning results when toxicant concentrations in an organism reach a level that is toxic to the organisms that feed on it. Substances that bioaccumulate or biomagnify in food webs often are considered to have the greatest potential to cause secondary poisoning. Biomagnification is the process by which tissue concentrations of a bioaccumulated substance increase as it is passed up the food web through at least two trophic levels. Polychlorinated biphenyls (PCBs) and the organochlorine pesticide DDT are common examples of chemicals that biomagnify in food webs (Eisler 1986, Keith 1996). Woodwell et al. (1967), for example, observed that DDT concentrations were found to increase in step-wise fashion from one trophic level to the next, and measured DDT residues in birds were approximately one million times greater than the concentrations in water. The concern with these types of chemicals is that seemingly low environmental concentrations can have population-level effects in organisms in the upper levels of food webs. As discussed in this section, however, there is little evidence to suggest that metals[1] biomagnify in aquatic food webs. In addition, many metals do not bioaccumulate in aquatic food webs (i.e., tissue concentrations of some metals decrease with increasing trophic level). However, the scientific literature is somewhat contradictory on whether metals tend to result in secondary poisoning. Some studies suggest that metals can be quite toxic via food chain transfer (Dallinger et al. 1987, Woodward et al. 1994),

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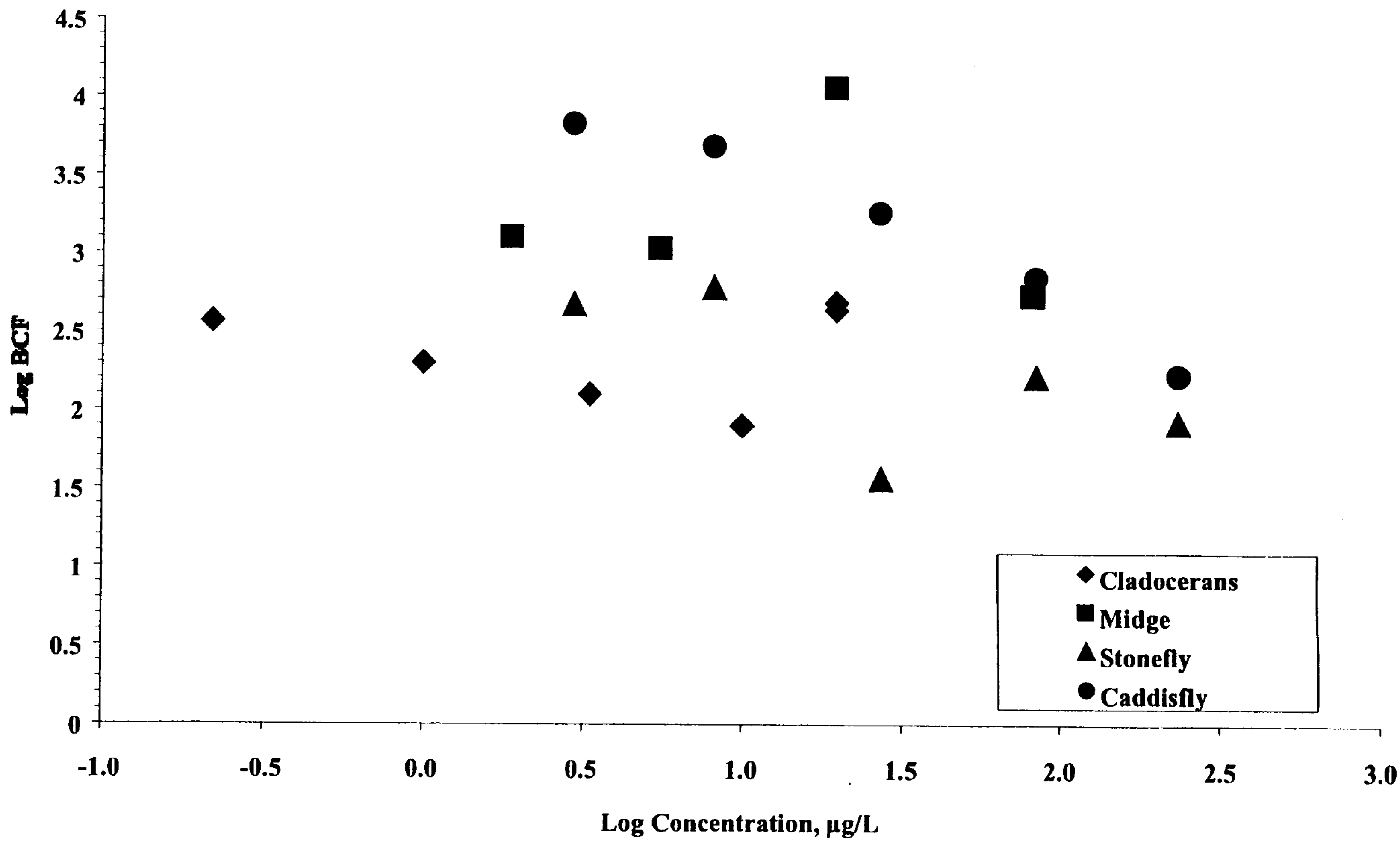
[1] This point refers to inorganic metal compounds, and not organometallic compounds such as methyl mercury.

Figure 24. Relationship Between Cadmium BCFs for Daphnids and Insects.



while others suggest that the diet contributes negligibly to metal toxicity (Macek et al. 1979, Hansen and Lambert 1987). With the exception of perhaps mercury and selenium, secondary poisoning by metals may largely be a site-specific issue.

Leland and Kuwabara (1985) state that the classic idea of biomagnification, developed from studies of DDT, does not hold for most metals. Macek et al. (1979) went a step further and questioned the importance of the food web as a pathway for accumulation of most chemicals. According to Macek et al. (1979), early investigators stressing the importance of food web transfer of chemicals were generally basing their hypotheses on studies with DDT. In addition, some researchers demonstrating the importance of food web transfer failed to factor in the quantitative significance of different exposure pathways. Macek et al. (1979) studied cadmium and several organic chemicals to determine the importance of the dietary contribution of a chemical to the body burden. The dietary contribution was only substantial for DDT, while for cadmium, the diet contributed to only 1.2 percent of the body burden in shrimp.

The limited bioavailability of inorganic forms of metals in food may explain why metals are generally not considered secondary poisons. It is generally assumed that higher residues of trace substances in the food chain should result in greater hazards to the consumer; however, studies have shown that there are a number of modifying factors that reduce the potential for adverse effects (Hansen and Lambert 1987). Absorption of metals from food is highly variable because of the variety of free and bound forms of the ions that are possible in food (Spacie and Hamelink 1985). In addition, competition between related elements for active transport sites is also variable. The following further discusses food chain transfer of metals and whether it is likely to result in secondary poisoning. For comparison, food chain transfer and secondary poisoning by certain organometallic and organic compounds are also discussed.

2.5.1 Metals and Inorganic Metallic Compounds

2.5.1.1 Cadmium

According to Suedel et al. (1994), there is little evidence to suggest that cadmium biomagnifies in aquatic systems. Ferard et al. (1983) examined the transfer of cadmium in an experimental food chain consisting of algae (*Chlorella vulgaris*), zooplankton (*Daphnia magna*), and fish (*Leucaspias delineatus*). Algae were exposed to one of four cadmium concentrations for 10 days, *Daphnia* were allowed to feed on the exposed algae for 20 days, and fish were allowed to feed on the *Daphnia* for 4 days. Algae exposed to concentrations of 10, 50, 100, and 250 µg/L accumulated cadmium to concentrations of 30, 92, 210, and 570 mg/kg dw, respectively. Cadmium concentrations in *Daphnia* were similar to or less than those in the algae they fed upon – 32, 44, 58, and 259 mg/kg dw at successively higher exposure concentrations. Despite the lower cadmium residues, however, *Daphnia* reproduction was impaired at all exposure levels. In the fish, cadmium concentrations were less than one mg/kg dw at all exposure levels. These results demonstrate that cadmium does not biomagnify and they suggest that cadmium concentrations may decrease with increasing trophic level, as demonstrated by cadmium concentrations in the algae and *Daphnia*. The even lower concentrations measured in the fish may simply be a function of the short exposure time over which fish were allowed to feed on contaminated *Daphnia*. The main point of this study is that cadmium concentrations appear to decrease with increasing trophic level.

It should be noted that the toxicity in *Daphnia* observed by Ferard et al. (1983) does not necessarily imply that cadmium should be regarded as a secondary poison since the original cadmium concentrations to which algae were exposed are much greater than what would be considered environmentally relevant. Furthermore, cadmium concentrations in water were not measured, although cadmium was likely released from the algae into the water. Consequently, the observed toxicity in *Daphnia* may have been from a waterborne exposure. Further studies should be conducted at environmentally relevant concentrations to confirm whether it should be of concern.

As summarized by Rainbow (1989), certain pelagic seabirds contain extremely high concentrations of cadmium in their kidney and liver, and these levels appear to be natural in origin. A probable source of the cadmium is the diet and as a result, Rainbow (1989) evaluated cadmium concentrations in two Antarctic populations of the pelagic amphipod *Themisto gaudichaudii* and an Atlantic population of *T. compressa*. Cadmium concentrations in these amphipods are atypically high and do not appear to be of anthropogenic origin. In contrast to some studies, this study demonstrates that cadmium may be naturally bioaccumulated to high levels in some food webs and, as such, the potential for secondary poisoning with respect to hazard classification must be considered carefully.

There are a number of modifying factors that reduce the potential for biological effects due to metals in the diet. Cadmium, for example, interacts with many other nutritional elements and its bioavailability is influenced by diet, nutrition, and chemical species (Hansen and Lambert 1987). In short-term feeding studies, data indicate that cadmium bound to metallothionein is less bioavailable than cadmium salts (Hansen and Lambert 1987). This is consistent with the review of Spacie and Hamelink (1985) who suggest that the organically bound fraction of metal in food is relatively unavailable for uptake in the gut. Limited metal bioavailability in birds has also been observed. Intestinal uptake of cadmium in Japanese quail, for example, was dose-dependent and represented only about 0.4 to 2 percent of the dose (Furness 1996).

2.5.1.2 Copper

There is no evidence that copper biomagnifies in aquatic systems, although it does appear to be transferred through food chains (Suedel et al. 1994). As reviewed by Lewis and Cave (1982), copper accumulation in aquatic organisms at different trophic levels varies considerably and depends on several factors, including the physiological requirements of the organism, the source of copper, exposure duration, migration patterns, and chemical speciation.

No studies were identified that conclusively demonstrate copper results in secondary poisoning. Woodward et al. (1994) fed rainbow trout (*Oncorhynchus mykiss*) fry benthic invertebrates from the Clark Fork River, Montana for 91 days. The benthic invertebrates had elevated concentrations of copper, as well as elevated levels of arsenic, cadmium, and lead. This study demonstrated that dietary copper can be an important source for copper bioaccumulation in exposed fish, but it cannot be conclusively stated whether the observed toxicity was due to copper, since concentrations of other metals were elevated. The following discusses dietary toxicity studies in which fish were exposed to diets with only elevated copper concentrations.

Kamunde (1999) exposed rainbow trout to dietary copper concentrations of 11 (control), 300, and 1,000 mg/kg for 28 days. The results suggested that the gut appears to present a strong barrier to internal uptake of elevated dietary copper. Gut tissue concentrations increased slightly, but significantly, in fish fed 300 mg/kg copper and approximately an order of magnitude in fish fed 1,000 mg/kg copper. No effects were observed on growth of the fish at these dietary copper levels. It appears that elevated dietary copper levels alter the energy budget by expending more energy toward copper regulation, but a higher dietary intake rate counteracted these effects and the growth rates remained similar to the controls (Kamunde 1999).

As another example, Mount et al. (1994) fed rainbow trout fry enriched with copper for 60 days. Dietary copper concentrations of 660 and 800 mg/kg dw had no effect on fish growth, but resulted in approximately 30 percent mortality. These concentrations are higher than those measured in invertebrates in the Clark Fork River, Montana, a location with extremely elevated metals concentrations. The authors hypothesize that the observed toxicity may actually have been due to waterborne copper. If 20 percent of the dietary copper was lost to the water (20 µg/L), and added to the level of copper already in the water (23 µg/L), the waterborne copper concentration would reach a level that is acutely toxic to trout (Mount et al. 1994). Moreover, Mount et al. (1994) note that toxicity in other studies have not been observed at similar dietary levels, and that in one study, 30 percent mortality was not observed until the dietary copper concentration reached 3,088 mg/kg dw. As such, this study further supports that copper is unlikely to cause secondary poisoning at environmentally relevant concentrations. Miller et al. (1993) fed rainbow trout (*O. mykiss*) a synthetic trout diet containing either 13 or 684 mg/kg copper. The fish fed the 13 mg/kg diet were also exposed to aqueous copper concentration of either 5, 32, 55, or 106 µg/L, while fish fed the 684 mg/kg diet were exposed to aqueous copper concentrations of either 13, 38, 62, or 127 µg/L. The experiment was conducted for 42 days. None of these combinations of dietary and waterborne copper concentrations affected trout survival or growth.

2.5.1.3 Lead

According to reviews of Eisler (1988) and Suedel et al. (1994), there is no evidence that lead biomagnifies in higher trophic levels of either freshwater or marine food webs. As reviewed by Demayo et al. (1982), dietary lead may be virtually unavailable to fish such as rainbow trout. This is supported by the studies summarized below.

Simulating an aquatic food chain in the laboratory, Vighi (1981) exposed *Selenastrum capricornutum* (green alga), *Daphnia magna* (zooplankton), and *Poecilia reticulata* (guppy) to lead nitrate for four weeks. Lead concentrations in guppies were three to four times greater in fish exposed to lead via water and food than in fish exposed to lead via water only. However, lead residues were still low (3.2 to 7.2 mg/kg ww, assuming a moisture content of 80 percent) and found to decrease with increasing trophic level. Given that lead residues were low in fish and found to decrease with increasing exposure level, this study provides evidence to suggest that lead should not be of concern as a secondary poison.

In a field study, Henny et al. (1991) similarly observed decreasing lead concentrations with increasing trophic level. Henny et al. evaluated fish and ospreys in a portion of the Coeur d'Alene River (Idaho) contaminated with high levels of lead. Whole fish collected along the river had elevated lead concentrations compared to fish collected at intermediate or reference areas. Mean whole body lead concentrations in fish from the river ranged from 0.75 mg/kg ww in largemouth bass (*Micropterus salmoides*) to 21.6 mg/kg ww in brown bullhead (*Ictalurus nebulosus*). Blood levels in ospreys were inversely related to δ-aminolevulinic acid dehydratase (ALAD) activity, indicating that they had recently been exposed to lead (decreased ALAD activity is an indicator of exposure, but not of effects). However, lead levels in ospreys were lower than in fish, presumably because bones and hard parts of the fish are not ingested and this is where lead tends to accumulate (Henny et al. 1991). Moreover, despite elevated lead concentrations occurring in river sediments and bioaccumulating in fish, there was no evidence of secondary poisoning of lead through the food chain from sediments because no effects on osprey reproductive performance along the river were observed.

2.5.1.4 Nickel

There is also no evidence that nickel biomagnifies in aquatic food webs (Suedel et al. 1994). Watras et al. (1985) studied nickel accumulation in *Daphnia magna* fed nickel-enriched algae and non-enriched algae (*Scenedesmus obliques*) for 13 days. *Daphina* were exposed to nickel via algae only, water only, or a combination of algae and water. *Daphnia* exposed to nickel via algae only accumulated nickel to a concentration of only 29 µg/kg ww, while *Daphnia* exposed to nickel via water only accumulated a much higher nickel concentration of 681 µg/kg ww. There was no indication that these levels resulted in effects to *Daphnia*. This study, therefore, demonstrates that nickel is not transferred significantly between trophic levels (i.e., through ingestion), nor does it appear to result in secondary poisoning (at least at lower trophic levels). In a field study reported by Mathis and Cummings (1973), nickel concentrations were also found to decrease with increasing trophic level in a food web characterized by clams, oligochaetes, omnivorous fish, and carnivorous fish, again demonstrating that food chain transfer of nickel is minimal, as is its potential for secondary poisoning.

2.5.1.5 Zinc

As for the other metals discussed above, there is no evidence that zinc biomagnifies in aquatic systems (Suedel et al. 1994). Given that zinc is an essential element, many organisms are known to accumulate zinc to high levels. Elevated accumulation rates may sometimes be mistaken as trophic transfer (Suedel et al. 1994). As discussed above, it appears that the organically bound fraction of metal in food is relatively unavailable for uptake in the gut. This is also confirmed based on data for zinc. In the sunfish (*Lepomis gibbosus*), for example, more zinc was accumulated from an artificial diet than from a natural diet of snails containing the same levels of zinc (Merlini et al. 1976). No studies were identified that explicitly assessed the potential for zinc to cause secondary poisoning.

2.5.1.6 Metal Mixtures

According to Dallinger et al. (1987), metals can reach concentrations in tissues that result in secondary poisoning even if they do not bioaccumulate to high levels or biomagnify. Woodward et al. (1994) fed rainbow trout fry (*O. mykiss*) invertebrates from the Clark Fork River, Montana that had large concentrations of metals. The guts of the invertebrates were not purged, so the metal concentrations in the invertebrates were based on accumulated metal in tissue plus metals bound to sediment in the gut. The mean concentrations of aluminum, arsenic, cadmium, copper, lead, and zinc were 1,759, 43.1, 3.12, 381, 32.7, and 528 mg/kg dw, respectively. Fry were also simultaneously exposed to water solutions with non-detectable levels of metals, solutions simulating typical metal levels in the river - cadmium, copper, lead, and zinc concentrations were 1.1, 12, 3.2, and 50 µg/L, respectively, or double these aqueous concentrations. The fry fed invertebrates with high metals levels had significantly reduced survival and growth, regardless of the aqueous metal concentrations to which they were exposed.

In a second study with a similar study design, early life stages of rainbow trout (*O. mykiss*) and brown trout (*Salmo trutta*) were exposed to aqueous metals in simulated Clark Fork water and dietary metals from invertebrates collected from the river (Woodward et al. 1995). The aqueous metal concentrations were the same as those tested in Woodward et al. (1994), while metal concentrations in invertebrates were lower, with the exception of zinc. Specifically, mean arsenic, cadmium, copper, lead, and zinc concentrations in invertebrates were approximately 19, <0.26, 174, 15, and 648 mg/kg dw. Exposure to any combination of aqueous and dietary metal concentrations did not have any effects on survival; however, effects on growth were observed. In brown trout, for example, exposure to aqueous metals only resulted in a 25 percent reduction in growth relative to fish exposed to reference water and diet, a 40 percent reduction in fish exposed to dietary metals alone, and a 50 percent reduction in fish exposed to both aqueous and dietary metals. Similar reductions in growth were observed in rainbow trout exposed to both aqueous and dietary metals. The Woodward et al. (1994, 1995) studies suggest that metal residues can result in secondary poisoning.

Mount et al. (1994) exposed rainbow trout (*O. mykiss*) to live diets (*Artemia* sp.) enriched with arsenic, cadmium, copper, lead, and zinc. No effects were observed on survival, weight, or length after 60 days of exposure to any of the treatments. The highest treatment contained dietary arsenic, cadmium, copper, lead, and zinc concentrations of 63, 21, 250, 82, and 740 mg/kg dw, respectively. These results are inconsistent with those reported by Woodward et al. summarized above, in which the trout diets were based on field-exposed invertebrates.

The above studies provide evidence that metals may cause secondary poisoning, although the evidence is somewhat conflicting. As discussed above, organisms have been observed to tolerate much higher metals concentrations in the laboratory. The differences in sensitivities observed in the Woodward et al. studies and laboratory studies discussed above is unclear. It may be a function of the difference in bioavailabilities between naturally and artificially incorporated metals, but there even seems to be conflicting evidence on this point. Merlini et al. (1976) observed that sunfish accumulated more zinc from an artificial diet than from a natural one, while Harrison and Curtis (1992) reported that environmentally contaminated natural foods have a greater absorption efficiency that surficially contaminated artificial diets. This is different than for some organometallics (e.g., methyl mercury, and organoselenium) where secondary poisoning has been documented in both the field and laboratory.

2.5.2 Organometallic Compounds

As the name suggests, an organometallic compound consists of a metal and an attached organic moiety. Organometallic compounds may be formed naturally in the environment (e.g., methylmercury, organoselenium) or from anthropogenic sources (e.g., organolead, tributyltin). Organometallic compounds tend to have greater bioaccumulation potential than inorganic metallic compounds, and some are known to biomagnify. Some representative organometallic compounds are discussed below for comparison with inorganic metals.

2.5.2.1 Naturally-Derived Organometallics

Methylmercury

Methylmercury is produced naturally in the environment via microbial methylation of inorganic mercury (II) (Weiner and Spry 1996) and is known to biomagnify in aquatic food webs (Biddinger and Gloss 1984, Eisler 1987, Sadiq 1992, Watras and Bloom 1992, Mason et al. 1995). In fish, it is estimated that greater than 90 percent of the methylmercury accumulated is through the diet (Wiener and Spry 1996). The high assimilation efficiency of methylmercury in fish is probably greater than 65 to 80 percent, while the absorption rate in mice has been found to be even higher, approximately 98 percent (Clarkson 1971). Given its high assimilation efficiency in fish, methylmercury rapidly penetrates and is cleared from the gut, binds to red blood cells, and is rapidly transported to all organs. In addition to its high assimilation efficiency, methylmercury tends to biomagnify in aquatic food webs because it is eliminated very slowly in fish relative to its rate of uptake (McKim et al. 1976). Moreover, unlike many inorganic metals, including inorganic mercury, methylmercury does not induce metallothionein or bind to existing metallothionein with much affinity (Wiener and Spry 1996). Based on its biochemical properties, its propensity to biomagnify, and toxicological studies in the laboratory and field, mercury in its methylated form should be considered a secondary poison. **Organoselenium**

In lentic water bodies with high biological activity, selenium is transferred through the food web as organoselenium compounds. For example, organoselenium compounds can bioaccumulate to levels in aquatic invertebrates that are nontoxic to the invertebrates themselves, but toxic to the shorebirds that feed upon them (Ohlendorf et al. 1986). Similar to methyl mercury, absorption of organoselenium in the gut of experimental animals has been shown to be high (i.e., 95-97 percent in rats [Thomson and Stewart 1973]). There is no evidence, however, that organoselenium compounds biomagnify in the food web (Sandholm et al. 1973, Suedel et al. 1994). Besser et al. (1993) exposed a simulated food chain consisting of algae (*Chlamydomona reinhardtii*), daphnids (*Daphnia magna*), and bluegill sunfish to organic and inorganic selenium. Except at very low exposure concentrations, daphnids and bluegill did not accumulate selenium concentrations greater than those in their diet. However, organoselenium has been shown to be a secondary poison to fish and shorebirds (Finley et al. 1985, Ohlendorf et al. 1986).

Organic Arsenicals

The propensity for organic arsenic compounds to biomagnify or cause secondary poisoning in aquatic biota has not been extensively studied. It is generally considered that organic arsenic compounds do not biomagnify because they are easily excreted (Biddinger and Gloss 1984). Cockell and Hilton (1988) exposed juvenile rainbow trout (*O. mykiss*) to dietary organic arsenic (as dimethylarsinic acid or arsanilic acid) concentrations of 200, 400, 800, and 1,600 mg/kg for eight weeks. None of these dietary levels had any significant effects on growth or survival. The highest concentration tested exceeded 1,000 times the background level in the control diet (Cockell and Hilton 1988). The arsenic data are limited, but it is important to note that not all organometallic compounds behave similarly with regards to biomagnification potential and secondary poisoning. As such, they should be classified separately on a compound-by-compound basis.

2.5.2.2 Anthropogenically-Derived Organometallics

It is beyond the scope of this report to provide a detailed review on all organometallic compounds. The propensity for methyl mercury and organoselenium compounds to result in secondary poisoning was summarized in the previous section since it has been well document in both laboratory and field studies. Other organometallic compounds, particularly those of anthropogenic origin (e.g., tetraethyllead, organotins), have not been well studied in food webs. It should not be assumed that these compounds behave like methyl mercury or organoselenium in the environment. In addition, it should not be assumed that individual compounds within an organometallic group will behave the same. Tributyltin, for example, behaves much differently and has a much different toxicity potential than other organotins (e.g., monobutyltin, dibutyltin, octyltins). As a result, these compounds need further review before interpretation can be provided on their potential to bioaccumulate and result in secondary poisoning.

2.5.3 Conclusions on Secondary Poisoning

Limited bioavailability from dietary sources is probably the key parameter that explains why most conclude that inorganic forms of metals do not result in secondary poisoning. It has been demonstrated in both laboratory and field studies that concentrations of some metals decrease with increasing trophic level. Even though some aquatic biota may bioaccumulate metals to levels higher than those to which they were exposed, these metals tend to have limited bioavailability to the organisms that feed upon them. Because many organisms have developed strategies that allow for the naturally high bioaccumulation of some metals in nontoxic forms, it also suggests that there is a mechanism by which consumers are not poisoned by feeding on these organisms. Although no studies on this topic are known, it suggests that detoxified metal granules are probably not bioavailable to biota in higher trophic levels. As discussed above, some studies have suggested that metals bound to metallothioneins are also not bioavailable to upper trophic level biota.

The majority of the data indicate that inorganic metal compounds are not secondary poisons. A limited number of studies (e.g., Woodward et al. 1994, 1995) in which fish were fed field-exposed invertebrates indicate that secondary poisoning may be an issue in some site-specific situations. However, with any field study, or study with a field-based component, there may be unknown confounding factors that influence the results. The results have yet to be duplicated in laboratory-based food chains. On the other hand, the naturally-derived organometallics methylmercury and organoselenium have been documented in both field and laboratory studies to result in secondary poisoning.

3. CONCLUSIONS

As summarized by Chapman (1996), hazard classification of metals and metal compounds should be determined based on three specific questions:

1.) Is a substance bioavailable such that adverse environmental effects may occur?;

2.) If bioavailable, is a substance likely to cause short-term adverse effects to aquatic organisms?; and

3.) If bioavailable, but not exhibiting short-term adverse effects to aquatic organisms, is a substance likely to cause long-term adverse effects to aquatic organisms?

As demonstrated in this report, none of these questions are addressed by using BCFs for metals or inorganic metal compounds. Accordingly, two major questions were addressed in this report with regards to metals and metal compounds: (1) Is there a direct relationship between bioaccumulation potential and direct toxicity? In other words, do the more bioaccumualtive metals tend to be more toxic with increasing bioaccumulation potential?; and (2) Is there a relationship between bioaccumulation potential and secondary poisoning or biomagnification of metals? The answers to each of these questions are summarized below.

Is there a relationship between bioaccumulation potential and direct toxicity?

"It is like stating the obvious: trace metals can be accumulated by freshwater invertebrates (Timmermans 1993)." Invertebrates, and all aquatic organisms, must control intracellular trace metals concentrations that are essential for life. As such, they have developed various mechanisms that allow them to (1) acquire (i.e., accumulate) sufficient amounts of trace metals from low ambient conditions, and (2) regulate any influx of essential trace elements beyond metabolic requirements or non-essential trace elements that pose potential toxicity at low concentrations (Phillips and Rainbow 1989). The regulatory strategies of aquatic organisms range from active regulation (i.e., active excretion of excess metal) to storage (i.e., excess metals are stored in detoxified forms). For simplicity, these two extremes in regulatory strategies are discussed further in order to summarize the key points from this report.

Some organisms can regulate certain metals, such that all metal that is accumulated in excess of its metabolic requirements is excreted (Rainbow 1996). As the external metal concentration increases, the regulatory mechanism is overwhelmed and net accumulation occurs (i.e., the uptake rate exceeds the excretion rate). In organisms such as these, the bioaccumulation potential of metals cannot be described by individual BCFs. The organism is able to maintain a fairly constant tissue concentration of essential metals over a wide range of metal exposure concentrations. As a result, the BCF is dependent on the water concentration to which the organism is exposed – the bioaccumulation potential of a metal may appear extremely high at a low water concentration, but negligible at a high water concentration. In addition, because these organisms regulate metals through active uptake and excretion and do not have well developed mechanisms for storing excess metal, these organisms are unable to accumulate substantial amounts of metal before toxicity results.

At the other end of the spectrum, an organism may sequester all metal that is accumulated with no significant excretion (Rainbow 1996). For these organisms, the metal is stored in a detoxified form. Although such an organism may have a high BCF, it also has a storage mechanism that renders the metal non-toxic. Since metals are stored in non-toxic forms, the high bioaccumulation potential in these organisms, of course, has no relationship to the toxic potential of the metal.

Many studies are available in the scientific literature demonstrating the lack of a relationship between bioaccumulation potential and direct metal toxicity. This report also provided multiple analyses demonstrating the lack of a relationship between bioaccumulation and toxicity. For example, the range in BCFs for cadmium and zinc over multiple species is almost identical despite cadmium being many times more toxic than zinc. Lastly, analyses of species sensitivity distributions for multiple metals suggest that toxicity is partially a function of an organism's regulatory mechanism, not its bioaccumulation potential. In fact, based on the analyses provided in this report, it is more likely that an inverse relationship exists between bioaccumulation potential and direct toxicity.

Is there a relationship between bioaccumulation potential and secondary poisoning?

Secondary poisoning results when an organism is adversely affected by a substance that has accumulated in its food items. Substances that biomagnify or bioaccumulate in food webs often are considered to have the potential to cause secondary poisoning. For metals, however, there is little evidence to suggest that inorganic metals biomagnify in aquatic food webs. Leland and Kuwabara (1985) state that the classic idea of biomagnification is mainly developed from studies of DDT, but does not hold for most metals. In addition to the general lack of biomagnification potential for metals, it has been shown that several metals do not bioaccumulate appreciably in aquatic food webs. Cadmium, lead, and nickel concentrations in tissue have both been shown to decrease with increasing trophic level (Vighi 1981, Henny et al. 1991, Ferard et al. 1983, Mathis and Cummings 1973).

It is important to note, however, that although biomagnification of metals does not occur and the bioaccumulation potential of many metals in aquatic food webs is low. There is some field evidence that metal concentrations can be bioaccumulated to levels high enough to induce secondary poisoning (e.g., Woodward et al. 1994). As mentioned before, these studies have not been validated in the laboratory-based food chains to eliminate confounding factors. However, given these data, further research is necessary to resolve the importance of dietary exposure in assessing the hazard of inorganic metals and metal compounds.

In summary, bioaccumulation is not an appropriate parameter for hazard classification of metals and inorganic metal compounds because (1) many organisms naturally bioaccumulate metals to high levels, (2) it is not possible to estimate the bioaccumulation potential of metals in many organisms because BCFs tend to be dependent on exposure concentration, (3) bioaccumulation potential cannot be related to direct toxicity, and (4) bioaccumulation potential cannot be related to the potential for secondary poisoning. Given that bioaccumulation potential of metals cannot be related to direct toxicity or secondary poisoning, it should not be used as a parameter in hazard classification of metals.

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APPENDIX A

DATABASE OF BIOCONCENTRATION FACTORS (BCFs)

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Aluminum	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	1750000	175000	1800	97.2	Riley and Roth 1971
Aluminum	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	307000	30700	1800	17.1	Riley and Roth 1971
Aluminum	Algae (*Chlorella salina*)	Algae	SW	25	Cells	118000	11800	1800	6.6	Riley and Roth 1971
Aluminum	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	.289000	28900	1800	16.1	Riley and Roth 1971
Aluminum	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	160000	16000	1800	8.9	Riley and Roth 1971
Aluminum	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	550000	55000	1800	30.6	Riley and Roth 1971
Aluminum	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	169000	16900	1800	9.4	Riley and Roth 1971
Aluminum	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	735000	73500	1800	40.8	Riley and Roth 1971
Aluminum	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	225000	22500	1800	12.5	Riley and Roth 1971
Aluminum	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	275000	27500	1800	15.3	Riley and Roth 1971
Aluminum	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	358000	35800	1800	19.9	Riley and Roth 1971
Aluminum	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	490000	49000	1800	27.2	Riley and Roth 1971
Aluminum	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	478000	47800	1800	26.6	Riley and Roth 1971
Aluminum	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	255000	25500	1800	14.2	Riley and Roth 1971
Aluminum	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	425000	42500	1800	23.6	Riley and Roth 1971
Aluminum	Brook trout (*Salvelinus fontinalis*)	Fish	FW	30	WB	—	12000	242	50	Cleveland et al. 1986
Aluminum	Brook trout (*Salvelinus fontinalis*)	Fish	FW	30	WB	—	33000	242	136	Cleveland et al. 1986
Arsenic (III)	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	520	130	4.0	Barrows et al. 1980
Arsenic (III)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	21	WB	16400	3000	2950	1.0	Dixon and Sprague 1981
Arsenic (III)	Cladoceran (*Daphnia magna*)	Invert	FW	21	WB	48500	9700	970	10.0	Spehar et al. 1980
Arsenic (III)	Cladoceran (*Daphnia magna*)	Invert	FW	21	WB	21024	4204.8	96	43.8	Spehar et al. 1980
Arsenic (III)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	112	Soft parts	—	—	—	350	USEPA 1985a
Arsenic (III)	Snail (*Helisoma campanulatum*)	Invert	FW	28	WB	80000	8000	961	8.3	Spehar et al. 1980
Arsenic (III)	Snail (*Helisoma campanulatum*)	Invert	FW	28	WB	27000	2700	88	30.7	Spehar et al. 1980
Arsenic (III)	Snail (*Stagnicola emarginata*)	Invert	FW	28	WB	3100	310	88	3.5	Spehar et al. 1980
Arsenic (III)	Snail (*Stagnicola emarginata*)	Invert	FW	28	WB	15000	1500	961	1.6	Spehar et al. 1980
Arsenic (III)	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	1000	100	88	1.1	Spehar et al. 1980
Arsenic (III)	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	43000	4300	961	4.5	Spehar et al. 1980
Arsenic (V)	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	—	—	—	3	USEPA 1985a
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	200	10	20	McGeachy and Dixon 1990
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	300	1390	0.22	McGeachy and Dixon 1990
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	1000	16250	0.06	McGeachy and Dixon 1990
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	100	10	10	McGeachy and Dixon 1990
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	250	1440	0.17	McGeachy and Dixon 1990
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	800	8360	0.10	McGeachy and Dixon 1990
Arsenic (V)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	77	WB	—	1750	18050	0.10	McGeachy and Dixon 1990
Arsenic (V)	Cladoceran (*Daphnia magna*)	Invert	FW	21	WB	20000	4000	1000	4.0	Spehar et al. 1980
Arsenic (V)	Cladoceran (*Daphnia magna*)	Invert	FW	21	WB	50000	10000	100	100.0	Spehar et al. 1980
Arsenic (V)	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	28	WB	6120	1224	0.71	1723.9	Lindsay and Sanders 1990
Arsenic (V)	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	28	WB	5750	1150	9.67	118.9	Lindsay and Sanders 1990

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Arsenic (V)	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	28	WB	5160	1032	24.6	42.0	Lindsay and Sanders 1990
Arsenic (V)	Snail (*Helisoma campanulatum*)	Invert	FW	28	WB	8800	880	89	9.9	Spehar et al. 1980
Arsenic (V)	Snail (*Helisoma campanulatum*)	Invert	FW	28	WB	28000	2800	961	2.9	Spehar et al. 1980
Arsenic (V)	Snail (*Stagnicola emarginata*)	Invert	FW	28	WB	3100	310	88	3.5	Spehar et al. 1980
Arsenic (V)	Snail (*Stagnicola emarginata*)	Invert	FW	28	WB	14000	1400	961	1.5	Spehar et al. 1980
Arsenic (V)	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	12000	2400	91.6	26.2	Spehar et al. 1980
Arsenic (V)	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	31000	6200	973	6.4	Spehar et al. 1980
Beryllium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	5130	270	19.0	Barrows et al. 1980
Cadmium	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	20000	2000	75	26.7	Canterford et al. 1978
Cadmium	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	23000	2300	150	15.3	Canterford et al. 1978
Cadmium	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	24000	2400	300	8.0	Canterford et al. 1978
Cadmium	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	21000	2100	450	4.7	Canterford et al. 1978
Cadmium	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	50000	5000	600	8.3	Canterford et al. 1978
Cadmium	NW salamander (*Ambystoma gracile*)	Amphibian	FW	24	WB	—	1620	48.9	33.1	Nebeker et al. 1995
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	330	Gill	—	34000	31	1096.8	Eaton 1974
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	330	Kidney	—	188000	31	6064.5	Eaton 1974
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	330	Liver	—	201000	31	6483.9	Eaton 1974
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	40	0.03	1333.3	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	100	0.8	125.0	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	140	1.8	77.8	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	200	2.2	90.9	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	240	3.6	66.7	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	290	4.4	65.9	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	360	5.2	69.2	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	510	8.4	60.7	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	0	0.02	0.0	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	100	2.8	35.7	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	250	6.2	40.3	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	360	7.7	46.8	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	750	13.2	56.8	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	600	16.1	37.3	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	650	19.7	33.0	Cope et al. 1994
Cadmium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	1400	32.3	43.3	Cope et al. 1994
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW		Gill	600	120	0.06	2000.0	Benoit et al. 1976
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW		Gill	3000	600	0.5	1200.0	Benoit et al. 1976
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW		Kidney	1400	280	0.06	4666.7	Benoit et al. 1976
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW		Kidney	11000	2200	0.5	4400.0	Benoit et al. 1976
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW		Liver	300	60	0.06	1000.0	Benoit et al. 1976
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW		Liver	3000	600	0.5	1200.0	Benoit et al. 1976
Cadmium	Brook trout (*Salvelinus fontinalis*)	Fish	FW	93	Muscle	—	22	1	22.0	Sangalang and Freeman 1979

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	14	Gills	2200	440	27	16.3	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	90	Gills	8900	1780	27	65.9	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	14	Heart	30	6	27	0.2	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	90	Heart	0	0	27	0.0	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	14	Kidney	480	96	27	3.6	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	90	Kidney	5300	1060	27	39.3	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	14	Liver	470	94	27	3.5	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	90	Liver	5700	1140	27	42.2	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	14	Spleen	60	12	27	0.4	Roberts et al. 1979
Cadmium	Brown trout (*Salmo trutta*)	Fish	FW	90	Spleen	450	90	27	3.3	Roberts et al. 1979
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Backbone	126	25.2	5.44	4.6	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Dorsal fin	225	45	5.44	8.3	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Gills	2957	591.4	5.44	108.7	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Liver	1693	338.6	5.44	62.2	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Muscle (fillet)	177	35.4	5.44	6.5	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Otoliths	295	59	5.44	10.8	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Skin (dorsal)	290	58	5.44	10.7	Westernhagen et al. 1980
Cadmium	Dab (*Limanda limanda*)	Fish	SW	96	Skin (ventral)	632	126.4	5.44	23.2	Westernhagen et al. 1980
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Bile	—	400	130	3.1	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Digestive tract	—	6700	130	51.5	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Gill filaments	—	2000	130	15.4	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Kidneys	—	16000	130	123.1	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Liver	—	4800	130	36.9	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Muscles	—	200	130	1.5	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Skin	—	400	130	3.1	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	Spleen	—	1400	130	10.8	Noel-Lambot and Bouquegneau 1977
Cadmium	Eel (*Anguilla anguilla*)	Fish	SW	60	WB	—	560	130	4.3	Noel-Lambot and Bouquegneau 1977
Cadmium	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	200	40	0.1	400.0	Spehar et al. 1978
Cadmium	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	400	80	0.1	800.0	Spehar et al. 1978
Cadmium	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	40000	8000	4.3	1860.5	Spehar et al. 1978
Cadmium	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	16000	3200	6.3	507.9	Spehar et al. 1978
Cadmium	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	16000	3200	8.5	376.5	Spehar et al. 1978
Cadmium	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	28000	5600	8.5	658.8	Spehar et al. 1978
Cadmium	Guppy (*Poecilia reticulatal)*	Fish	FW	32	WB	—	12500	45	277.8	Canton and Slooff 1982
Cadmium	Lake charr (*Salvelinus namaycush*)	Fish	FW	109	Gill	—	1300	0.5	2600.0	Kislalioglu et al. 1996
Cadmium	Lake charr (*Salvelinus namaycush*)	Fish	FW	109	Gill	—	2130	5	426.0	Kislalioglu et al. 1996
Cadmium	Lake charr (*Salvelinus namaycush*)	Fish	FW	109	Kidney	—	1420	0.5	2840.0	Kislalioglu et al. 1996
Cadmium	Lake charr (*Salvelinus namaycush*)	Fish	FW	109	Kidney	—	3430	5	686.0	Kislalioglu et al. 1996
Cadmium	Lake charr (*Salvelinus namaycush*)	Fish	FW	109	Liver	—	1150	0.5	2300.0	Kislalioglu et al. 1996
Cadmium	Lake charr (*Salvelinus namaycush*)	Fish	FW	109	Liver	—	2800	5	560.0	Kislalioglu et al. 1996

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	14	Gills	29000	5800	9	644.4	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Gills	17700	3540	10	354.0	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	14	Heart	110	22	9	2.4	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Heart	100	20	10	2.0	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Kidney	—	379	10	37.9	Calamari et al. 1982
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Kidney	—	980	0.47	2085.1	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Kidney	—	16400	1.8	9111.1	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Kidney	—	52600	3.4	15470.6	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	560	Kidney	—	39800	5.5	7236.4	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	14	Kidney	510	102	9	11.3	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Kidney	7500	1500	10	150.0	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Liver	—	380	10	38.0	Calamari et al. 1982
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Liver	—	1100	0.47	2340.4	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Liver	—	5800	1.8	3222.2	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Liver	—	12300	3.4	3617.6	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	560	Liver	—	11100	5.5	2018.2	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	14	Liver	260	52	9	5.8	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Liver	5200	1040	10	104.0	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Muscle	—	43	10	4.3	Calamari et al. 1982
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Muscle	—	250	0.47	531.9	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Muscle	—	160	1.8	88.9	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	455	Muscle	—	620	3.4	182.4	Brown et al. 1994
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	14	Spleen	0	0	9	0.0	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Spleen	340	68	10	6.8	Roberts et al. 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	210	WB	440	120	0.01	12000.0	Kumada et al. 1973
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	210	WB	280	50	0.1	500.0	Kumada et al. 1973
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	210	WB	1400	380	1	380.0	Kumada et al. 1973
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	210	WB	1900	540	2.2	245.5	Kumada et al. 1973
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	210	WB	2800	960	4.8	200.0	Kumada et al. 1973
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	70	WB	—	130	4	32.5	Kumada et al. 1980
Cadmium	Stickleback (species not provided)	Fish	FW	7	Brain	1180	236	120	2.0	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Brain	610	122	1070	0.1	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gall bladder	4220	844	120	7.0	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gall bladder	5250	1050	1070	1.0	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gill	4590	918	120	7.7	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gill	14030	2806	1070	2.6	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gonad	2510	502	120	4.2	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gonad	1100	220	1070	0.2	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gut	8970	1794	120	15.0	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Gut	19780	3956	1070	3.7	Woodworth and Pascoe 1983

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Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Cadmium	Stickleback (species not provided)	Fish	FW	7	Kidney	4820	964	120	8.0	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Kidney	3170	634	1070	0.6	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Liver	9030	1806	120	15.1	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Liver	25680	5136	1070	4.8	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Muscle	2280	456	120	3.8	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Muscle	930	186	1070	0.2	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Skin	3280	656	120	5.5	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Skin	5460	1092	1070	1.0	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Spleen	3420	684	120	5.7	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	Spleen	4940	988	1070	0.9	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	WB	300	60	120	0.5	Woodworth and Pascoe 1983
Cadmium	Stickleback (species not provided)	Fish	FW	7	WB	750	150	1070	0.1	Woodworth and Pascoe 1983
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	10	WB	3050	610	0.25	2440.0	Stephenson and Mackie 1989
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	10	WB	1700	340	0.05	6800.0	Stephenson and Mackie 1989
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	10	WB	1400	280	0.01	28000.0	Stephenson and Mackie 1989
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	10	WB	3600	720	0.25	2880.0	Stephenson and Mackie 1989
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	10	WB	1900	380	0.05	7600.0	Stephenson and Mackie 1989
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	10	WB	1500	300	0.01	30000.0	Stephenson and Mackie 1989
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	42	WB	23000	4600	<0.18	25555.6	Borgmann et al. 1991
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	42	WB	32000	6400	2.28	2807.0	Borgmann et al. 1991
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	42	WB	23000	4600	8.98	512.2	Borgmann et al. 1991
Cadmium	Amphipod (*Hyalella azteca*)	Invert	FW	42	WB	42000	8400	0.38	22105.3	Borgmann et al. 1991
Cadmium	Asiatic clam (*Corbicula fluminea*)	Invert	FW	28	Soft parts	75450	7545	23	328.0	Graney et al. 1983
Cadmium	Asiatic clam (*Corbicula fluminea*)	Invert	FW	28	Soft parts	95450	9545	55	173.5	Graney et al. 1983
Cadmium	Bay scallop (*Argopecten irradians*)	Invert	SW	42	Soft parts	1225000	122500	70	1750.0	Pesch and Stewart 1980
Cadmium	Bay scallop (*Argopecten irradians*)	Invert	SW	42	Soft parts	1745000	174500	130	1342.3	Pesch and Stewart 1980
Cadmium	Beetle (Dytiscidae)	Invert	FW		WB	—	—	—	164	USEPA 1985b
Cadmium	Beetle (Dytiscidae)	Invert	FW		WB	—	—	—	260	USEPA 1985b
Cadmium	Biting midge (Ceratopogonidae)	Invert	FW		WB	—	—	—	936	USEPA 1985b
Cadmium	Biting midge (Ceratopogonidae)	Invert	FW		WB	—	—	—	662	USEPA 1985b
Cadmium	Blue mussel (*Mytilus edulis*)	Invert	SW		Soft parts	—	—	—	113	USEPA 1985b
Cadmium	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	40519	5470	10	547.0	Phillips 1976
Cadmium	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	50519	6820	20	341.0	Phillips 1976
Cadmium	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	44593	6020	20	301.0	Phillips 1976
Cadmium	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	49481	6680	40	167.0	Phillips 1976
Cadmium	Caddisfly (*Hydropsyche betteni*)	Invert	FW	28	WB	100000	20000	3	6666.7	Spehar et al. 1978
Cadmium	Caddisfly (*Hydropsyche betteni*)	Invert	FW	28	WB	200000	40000	8.3	4819.3	Spehar et al. 1978
Cadmium	Caddisfly (*Hydropsyche betteni*)	Invert	FW	28	WB	250000	50000	27.5	1818.2	Spehar et al. 1978
Cadmium	Caddisfly (*Hydropsyche betteni*)	Invert	FW	28	WB	300000	60000	85.5	701.8	Spehar et al. 1978
Cadmium	Caddisfly (*Hydropsyche betteni*)	Invert	FW	28	WB	200000	40000	238	168.1	Spehar et al. 1978

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Cadmium	Cladoceran (*Daphnia magna*)	Invert	FW	4	WB	400	80	0.22	363.6	Poldoski 1979
Cadmium	Cladoceran (*Daphnia magna*)	Invert	FW	4	WB	1000	200	1.01	198.0	Poldoski 1979
Cadmium	Cladoceran (*Daphnia magna*)	Invert	FW	4	WB	2100	420	3.37	124.6	Poldoski 1979
Cadmium	Cladoceran (*Daphnia magna*)	Invert	FW	4	WB	4000	800	10.12	79.1	Poldoski 1979
Cadmium	Cladoceran (*Daphnia magna*)	Invert	FW	7	WB	48400	9680	20	484.0	Winner 1984
Cadmium	Cladoceran (*Daphnia magna*)	Invert	FW	7	WB	43300	8660	20	433.0	Winner 1984
Cadmium	Crayfish (*Cambarus latimanus*)	Invert	FW	150	WB	21960	7394	10	739.4	Thorp et al. 1979
Cadmium	Crayfish (*Orconectes propinquus*)	Invert	FW	8	WB	25000	5000	10	500.0	Gillespie et al. 1977
Cadmium	Crayfish (*Orconectes propinquus*)	Invert	FW	8	WB	140000	28000	100	280.0	Gillespie et al. 1977
Cadmium	Crayfish (*Orconectes propinquus*)	Invert	FW	8	WB	540000	108000	1000	108.0	Gillespie et al. 1977
Cadmium	Crayfish (*Oronectes virilis*)	Invert	FW	14	Abdominal muscle	3900	780	400	2.0	Mirenda 1986b
Cadmium	Crayfish (*Oronectes virilis*)	Invert	FW	14	Antennal Gland	39900	7980	400	20.0	Mirenda 1986b
Cadmium	Crayfish (*Oronectes virilis*)	Invert	FW	14	Carapace	4900	980	400	2.5	Mirenda 1986b
Cadmium	Crayfish (*Oronectes virilis*)	Invert	FW	14	Gill	225300	45060	400	112.7	Mirenda 1986b
Cadmium	Crayfish (*Oronectes virilis*)	Invert	FW	14	Hepatopancreas	90700	18140	400	45.4	Mirenda 1986b
Cadmium	Crayfish (*Oronectes virilis*)	Invert	FW	14	WB	28400	5680	400	14.2	Mirenda 1986b
Cadmium	Damselfly (*Ischnura* sp.)	Invert	FW		WB	—	—	—	1,300	USEPA 1985b
Cadmium	Damselfly (*Ischnura* sp.)	Invert	FW		WB	—	—	—	928	USEPA 1985b
Cadmium	Dragonfly (*Pantala hymenea*)	Invert	FW		WB	—	—	—	736	USEPA 1985b
Cadmium	Dragonfly (*Pantala hymenea*)	Invert	FW		WB	—	—	—	680	USEPA 1985b
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW		Soft parts	—	—	—	2,150	USEPA 1985b
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW		Soft parts	—	—	—	1,830	USEPA 1985b
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	119	Soft parts	—	106700	100	1067.0	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	112	Soft parts	—	105700	100	1057.0	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	91	Soft parts	—	96500	200	482.5	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	91	Soft parts	—	125800	200	629.0	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	119	Soft parts	—	106700	100	1067.0	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	112	Soft parts	—	105700	100	1057.0	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	91	Soft parts	—	96500	200	482.5	Shuster and Pringle 1969
Cadmium	Eastern oyster (*Crassostrea virginica*)	Invert	SW	91	Soft parts	—	125800	200	629.0	Shuster and Pringle 1969
Cadmium	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	42	WB	13400	2680	0.08	33500.0	Pesch and Stewart 1980
Cadmium	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	42	WB	20700	4140	0.14	29571.4	Pesch and Stewart 1980
Cadmium	Grass shrimp (*Palaemonetes vulgaris*)	Invert	SW	28	WB	—	20000	7.9	2531.6	Nimmo et al. 1977
Cadmium	Grass shrimp (*Palaemonetes vulgaris*)	Invert	SW	28	WB	—	3200	12.7	252.0	Nimmo et al. 1977
Cadmium	Grass shrimp (*Palaemonetes vulgaris*)	Invert	SW	28	WB	—	5000	28.2	177.3	Nimmo et al. 1977
Cadmium	Grass shrimp (*Palaemonetes vulgaris*)	Invert	SW	28	WB	—	6000	35.9	167.1	Nimmo et al. 1977
Cadmium	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	35	WB	—	8000	54	148.1	Nimmo et al. 1977
Cadmium	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	35	WB	—	13000	83	156.6	Nimmo et al. 1977
Cadmium	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	21	WB	35000	7000	50	140.0	Vernberg et al. 1977
Cadmium	Grass shrimp (*Palaemonetes pugio*)	Invert	SW	21	WB	20000	4000	50	80.0	Vernberg et al. 1977

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Cadmium	Mayfly (Ephemeroptera)	Invert	FW		WB	—	—	—	1,630	USEPA 1985b
Cadmium	Mayfly (Ephemeroptera)	Invert	FW		WB	—	—	—	3,520	USEPA 1985b
Cadmium	Midge (Chironomidae)	Invert	FW		WB	—	—	—	2,200	USEPA 1985b
Cadmium	Midge (Chironomidae)	Invert	FW		WB	—	—	—	1,830	USEPA 1985b
Cadmium	Midge (*Chironomus riparius*)	Invert	FW	30	WB	—	230000	20	11500.0	Timmermans et al. 1992
Cadmium	Midge (*Chironomus riparius*)	Invert	FW	) generation:	WB	12000	2400	1.9	1263.2	Postma and Davids 1995
Cadmium	Midge (*Chironomus riparius*)	Invert	FW	28-56	WB	30000	6000	5.6	1071.4	Postma and Davids 1995
Cadmium	Midge (*Chironomus riparius*)	Invert	FW	15	WB	224800	44960	83.4	539.1	Postma et al. 1996
Cadmium	Mussel (*Elliptio complanata*)	Invert	FW	3	Soft tissue	21000	2100	120	17.5	Wang and Evans 1993
Cadmium	Mussel (*Elliptio complanata*)	Invert	FW	3	Soft tissue	16000	1600	120	13.3	Wang and Evans 1993
Cadmium	Mussel (*Elliptio complanata*)	Invert	FW	3	Soft tissue	9500	950	120	7.9	Wang and Evans 1993
Cadmium	Mussel (*Elliptio complanata*)	Invert	FW	3	Soft tissue	8000	800	120	6.7	Wang and Evans 1993
Cadmium	Mussel (*Elliptio complanata*)	Invert	FW	3	Soft tissue	6000	600	120	5.0	Wang and Evans 1993
Cadmium	Mussel (*Mytilus edulis*)	Invert	SW	17	Soft tissue	10000	1000	8.4	119.0	Poulsen et al. 1982
Cadmium	Mussel (*Mytilus edulis*)	Invert	SW	17	Soft tissue	150000	15000	109	137.6	Poulsen et al. 1982
Cadmium	Oyster (*Crassostrea virginica*)	Invert	SW	280	Soft tissue	104910	13570	5	2714.0	Zaroogian and Cheer 1976
Cadmium	Oyster (*Crassostrea virginica*)	Invert	SW	252	Soft tissue	89000	8900	5	1780.0	Zaroogian 1980
Cadmium	Oyster (*Crassostrea virginica*)	Invert	SW	252	Soft tissue	176000	17600	10	1760.0	Zaroogian 1980
Cadmium	Oyster (*Crassostrea virginica*)	Invert	SW	252	Soft tissue	292000	29200	15	1946.7	Zaroogian 1980
Cadmium	Oyster (*Crassostrea virginica*)	Invert	SW	259	Soft tissue	91000	9100	5	1820.0	Zaroogian and Morrison 1981
Cadmium	Oyster (*Crassostrea virginica*)	Invert	SW	259	Soft tissue	270000	27000	15	1800.0	Zaroogian and Morrison 1981
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	30	Muscle	—	3800	79	48.1	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	30	Muscle	—	10400	182	57.1	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	30	Muscle	—	17000	307	55.4	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	30	Muscle	—	19400	586	33.1	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	30	Muscle	—	30100	866	34.8	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	30	Muscle	—	30500	1285	23.7	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	14	Gill	—	51600	224	230.4	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	14	Gill	—	60300	515	117.1	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	14	Gill	—	149800	736	203.5	Nimmo et al. 1977
Cadmium	Pink shrimp (*Penaeus duorarum*)	Invert	SW	14	Gill	—	176500	1010	174.8	Nimmo et al. 1977
Cadmium	Polychaete (*Ophryotrocha diadema*)	Invert	SW	64	WB	1708000	341600	1000	341.6	Klockner 1979
Cadmium	Polychaete (*Ophryotrocha diadema*)	Invert	SW	64	WB	315000	63000	100	630.0	Klockner 1979
Cadmium	Polychaete (*Ophryotrocha diadema*)	Invert	SW	64	WB	7900	1580	10	158.0	Klockner 1979
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	8	Gill	760	152	2	76.0	Zelikoff et al. 1995
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	17	Gill	850	170	2	85.0	Zelikoff et al. 1995
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	30	Gill	550	110	2	55.0	Zelikoff et al. 1995
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	17	Liver	250	50	2	25.0	Zelikoff et al. 1995
Cadmium	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	30	Liver	3000	600	2	300.0	Zelikoff et al. 1995
Cadmium	Snail (*Physa integra*)	Invert	FW	28	WB	30000	6000	3	2000.0	Spehar et al. 1978

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Cadmium	Snail (*Physa integra*)	Invert	FW	28	WB	50000	10000	8	1204.8	Spehar et al. 1978
Cadmium	Snail (*Physa integra*)	Invert	FW	28	WB	150000	30000	28	1090.9	Spehar et al. 1978
Cadmium	Soft-shell clam (*Mya arenaria*)	Invert	SW	70	Soft parts	—	8000	50	160.0	Pringle et al. 1968
Cadmium	Soft-shell clam (*Mya arenaria*)	Invert	SW	70	Soft parts	—	9000	100	90.0	Pringle et al. 1968
Cadmium	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	7000	1400	3	466.7	Spehar et al. 1978
Cadmium	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	25000	5000	8.3	602.4	Spehar et al. 1978
Cadmium	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	5000	1000	27.5	36.4	Spehar et al. 1978
Cadmium	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	70000	14000	85.5	163.7	Spehar et al. 1978
Cadmium	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	100000	20000	238	84.0	Spehar et al. 1978
Cadmium	Sydney Rock Oyster (*Saccostrea commercialis*)	Invert	SW	112	WB	—	—	9.8	12561	Ward 1982
Cadmium	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	2100	210	0.2	1050.0	Kraak et al. 1992
Cadmium	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	1100	110	0.2	550.0	Kraak et al. 1992
Cadmium	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	70000	7000	9	777.8	Kraak et al. 1992
Cadmium	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	300000	30000	42	714.3	Kraak et al. 1992
Cadmium	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	600000	60000	104	576.9	Kraak et al. 1992
Cadmium	Aquatic moss (*Rhynchostegium riparioides*)	Plant	FW	27	WB	530000	53000	43.6	1215.6	Mersch et al. 1993
Cadmium	Duck weed (*Lemna valdiviana*)	Plant	FW	21	WB	—	—	—	603	USEPA 1985b
Cadmium	Fern (*Salvinia natans*)	Plant	FW	21	WB	—	—	—	960	USEPA 1985b
Chromium (III)	Blue mussel (*Mytilus edulis*)	Invert	SW	42	Soft parts	430000	43000	1000	43.0	McDowell-Capuzzo and Sasner 1977
Chromium (III)	Soft-shell clam (*Mya arenaria*)	Invert	SW	42	Soft parts	765000	76500	1000	76.5	McDowell-Capuzzo and Sasner 1977
Chromium (III)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	6280	50	125.6	Shuster and Pringle 1969
Chromium (III)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	5810	50	116.2	Shuster and Pringle 1969
Chromium (III)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	11490	100	114.9	Shuster and Pringle 1969
Chromium (III)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	10870	100	108.7	Shuster and Pringle 1969
Chromium (total)	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	5500	550	12	45.8	Riley and Roth 1971
Chromium (total)	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	4400	440	12	36.7	Riley and Roth 1971
Chromium (total)	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	8400	840	12	70.0	Riley and Roth 1971
Chromium (total)	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	3600	360	12	30.0	Riley and Roth 1971
Chromium (total)	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	30000	3000	12	250.0	Riley and Roth 1971
Chromium (total)	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	4800	480	12	40.0	Riley and Roth 1971
Chromium (total)	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	13500	1350	12	112.5	Riley and Roth 1971
Chromium (total)	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	9400	940	12	78.3	Riley and Roth 1971
Chromium (total)	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	14700	1470	12	122.5	Riley and Roth 1971
Chromium (total)	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	4400	440	12	36.7	Riley and Roth 1971
Chromium (total)	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	8200	820	12	68.3	Riley and Roth 1971
Chromium (total)	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	2800	280	12	23.3	Riley and Roth 1971
Chromium (total)	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	8000	800	12	66.7	Riley and Roth 1971
Chromium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Kidney	—	3481	200	17.4	Calamari et al. 1982
Chromium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Liver	—	1980	200	9.9	Calamari et al. 1982
Chromium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	22	Muscle	—	340	2500	0.1	Buhler et al. 1977

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Chromium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Muscle	—	583	200	2.9	Calamari et al. 1982
Chromium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	24	WB	13	2.6	10	0.3	Fromm and Stokes 1962
Chromium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	24	WB	1.6	0.32	1.3	0.2	Fromm and Stokes 1962
Chromium (VI)	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	25000	5000	62	80.6	Ahsanulla and Williams 1991
Chromium (VI)	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	26000	5200	64	81.3	Ahsanulla and Williams 1991
Chromium (VI)	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	34000	6800	103	66.0	Ahsanulla and Williams 1991
Chromium (VI)	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	32000	6400	104	61.5	Ahsanulla and Williams 1991
Chromium (VI)	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	47000	9400	255	36.9	Ahsanulla and Williams 1991
Chromium (VI)	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	45000	9000	247	36.4	Ahsanulla and Williams 1991
Chromium (VI)	Blue mussel (*Mytilus edulis*)	Invert	SW		Soft parts	—	—	—	192	USEPA 1985c
Chromium (VI)	Eastern oyster (*Crassostrea virginica*)	Invert	SW		Soft parts	—	—	—	125	USEPA 1985c
Chromium (VI)	Polychaete (*Neanthes arenaceodentata*)	Invert	SW	163.3	WB	22201.005	4418	16.6	266.1	Oshida and Word 1982
Cobalt	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	9300	1860	14.7	127	Lind et al. Manuscript
Cobalt	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	14300	2860	29.5	97	Lind et al. Manuscript
Cobalt	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	21300	4260	48.7	87	Lind et al. Manuscript
Copper	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	105000	10500	34	308.8	Riley and Roth 1971
Copper	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	46000	4600	34	135.3	Riley and Roth 1971
Copper	Algae (*Chlorella salina*)	Algae	SW	25	Cells	25000	2500	34	73.5	Riley and Roth 1971
Copper	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	52000	5200	34	152.9	Riley and Roth 1971
Copper	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	57000	5700	34	167.6	Riley and Roth 1971
Copper	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	188000	18800	34	552.9	Riley and Roth 1971
Copper	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	93000	9300	34	273.5	Riley and Roth 1971
Copper	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	210000	21000	34	617.6	Riley and Roth 1971
Copper	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	95000	9500	34	279.4	Riley and Roth 1971
Copper	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	47000	4700	34	138.2	Riley and Roth 1971
Copper	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	62000	6200	34	182.4	Riley and Roth 1971
Copper	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	110000	11000	34	323.5	Riley and Roth 1971
Copper	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	29000	2900	34	85.3	Riley and Roth 1971
Copper	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	53000	5300	34	155.9	Riley and Roth 1971
Copper	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	90000	9000	34	264.7	Riley and Roth 1971
Copper	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	60000	6000	10	600.0	Canterford et al. 1978
Copper	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	25000	2500	20	125.0	Canterford et al. 1978
Copper	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	40000	4000	50	80.0	Canterford et al. 1978
Copper	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	70000	7000	80	87.5	Canterford et al. 1978
Copper	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	71000	7100	100	71.0	Canterford et al. 1978
Copper	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	60000	6000	150	40.0	Canterford et al. 1978
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Brain	29000	5800	3	1933.3	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Gill	3000	600	3	200.0	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Gill	3000	600	12	50.0	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Gill	3000	600	21	28.6	Benoit 1975

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Gill	5000	1000	40	25.0	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Gill	6000	1200	77	15.6	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Gonad	5000	1000	3	333.3	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Kidney	22000	4400	3	1466.7	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Kidney	9000	1800	12	150.0	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Kidney	12000	2400	21	114.3	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Kidney	13000	2600	40	65.0	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Kidney	12000	2400	77	31.2	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Liver	7000	1400	3	466.7	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Liver	8000	1600	12	133.3	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Liver	10000	2000	21	95.2	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Liver	61000	12200	40	305.0	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Liver	57000	11400	77	148.1	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Muscle	1000	200	3	66.7	Benoit 1975
Copper	Bluegill (*Lepomis macrochirus*)	Fish	FW		Spleen	18000	3600	3	1200.0	Benoit 1975
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Gill	3800	760	3.4	223.5	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Gill	4300	860	6.5	132.3	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Gill	4300	860	10	86.0	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Gill	3600	720	16	45.0	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Gill	6900	1380	27	51.1	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Gill	25000	5000	51	98.0	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Kidney	7800	1560	3.4	458.8	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Kidney	8500	1700	6.5	261.5	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Kidney	8800	1760	10	176.0	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Kidney	14300	2860	16	178.8	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Kidney	10000	2000	27	74.1	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Liver	29000	5800	3.4	1705.9	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Liver	28000	5600	6.5	861.5	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Liver	32000	6400	10	640.0	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Liver	47000	9400	16	587.5	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Liver	61000	12200	27	451.9	Brungs et al. 1973
Copper	Brown bullhead (*Ictalurus nebulosus*)	Fish	FW		Liver	130000	26000	51	509.8	Brungs et al. 1973
Copper	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	11600	2320	5	464	Lind et al. Manuscript
Copper	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	19800	3960	9	440	Lind et al. Manuscript
Copper	Stone loach (*Noemacheilus barbatulus*)	Fish	FW	63	Liver	77000	15400	120	128.3	Solbe and Cooper 1976
Copper	Stone loach (*Noemacheilus barbatulus*)	Fish	FW	63	Muscle	10100	2020	120	16.8	Solbe and Cooper 1976
Copper	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	540000	108000	10	10800.0	Ahsanulla and Williams 1991
Copper	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	79000	15800	3.5	4514.3	Borgmann et al. 1993
Copper	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	91000	18200	7.7	2363.6	Borgmann et al. 1993
Copper	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	92000	18400	10.7	1719.6	Borgmann et al. 1993

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Copper	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	95000	19000	16.7	1137.7	Borgmann et al. 1993
Copper	Amphipod (*Hyalella azteca*)	Invert	FW	28	WB	111000	22200	50.2	442.2	Borgmann and Norwood 1995
Copper	Amphipod (*Hyalella azteca*)	Invert	FW	42	WB	111000	22200	50.2	442.2	Borgmann and Norwood 1995
Copper	Asiatic clam (*Corbicula fluminae*)	Invert	FW	28	Soft parts	539350	53935	16	3370.9	Graney et al. 1983
Copper	Asiatic clam (*Corbicula fluminae*)	Invert	FW	28	Soft parts	774500	77450	57	1358.8	Graney et al. 1983
Copper	Bay scallop (*Argopecten irradians*)	Invert	SW	56	Soft parts	28000	2800	1.81	1547.0	Zaroogian and Johnson 1983
Copper	Bay scallop (*Argopecten irradians*)	Invert	SW	56	Soft parts	91000	9100	4.56	1995.6	Zaroogian and Johnson 1983
Copper	Bay scallop (*Argopecten irradians*)	Invert	SW	56	Soft parts	310000	31000	10.24	3027.3	Zaroogian and Johnson 1983
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	2593	350	10	35.0	Phillips 1976
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	10222	1380	20	69.0	Phillips 1976
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	54963	7420	40	185.5	Phillips 1976
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	3926	530	20	26.5	Phillips 1976
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	630	Soft parts	5510	551	1.0	551.0	Calabrese et al. 1984
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	630	Soft parts	19170	1917	5.0	383.4	Calabrese et al. 1984
Copper	Blue mussel (*Mytilus edulis*)	Invert	SW	630	Soft parts	62030	6203	10.0	620.3	Calabrese et al. 1984
Copper	Cladoceran (*Daphnia magna*)	Invert	FW	7	WB	70.7	14.14	30	0.5	Winner 1984
Copper	Cladoceran (*Daphnia magna*)	Invert	FW	7	WB	67.3	13.46	30	0.4	Winner 1984
Copper	Hard clam (*Mercenaria mercenaria*)	Invert	SW		–	—	—	—	88	USEPA 1985d
Copper	Soft-shell clam (*Mya arenaria*)	Invert	SW		–	—	—	—	3,300	USEPA 1985d
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Gill	35300	3530	7	504.3	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Gill	109500	10950	18	608.3	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Kidney	111100	11110	7	1587.1	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Kidney	27200	2720	18	151.1	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Muscle	34700	3470	7	495.7	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Muscle	35000	3500	18	194.4	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Viscera	57700	5770	7	824.3	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	Viscera	54500	5450	18	302.8	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	WB	47700	4770	7	681.4	Roesijadi 1980
Copper	Clam (*Protothaca staminea*)	Invert	SW	30	WB	46700	4670	18	259.4	Roesijadi 1980
Copper	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	694800	25	27792.0	Shuster and Pringle 1969
Copper	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	715600	25	28624.0	Shuster and Pringle 1969
Copper	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	1125000	50	22500.0	Shuster and Pringle 1969
Copper	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	943800	50	18876.0	Shuster and Pringle 1969
Copper	Isopod (*Asellus meridianus*)	Invert	FW	14	WB	800000	160000	500	320.0	Brown 1977
Copper	Isopod (*Asellus meridianus*)	Invert	FW	14	WB	800000	160000	500	320.0	Brown 1977
Copper	Isopod (*Asellus meridianus*)	Invert	FW	14	WB	2600000	520000	500	1040.0	Brown 1977
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	164140	45466.78	10	4546.7	McLusky and Phillips 1975
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	260160	72064.32	20	3603.2	McLusky and Phillips 1975
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	364650	101008.05	30	3366.9	McLusky and Phillips 1975
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	321430	89036.11	40	2225.9	McLusky and Phillips 1975

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	566530	156928.81	50	3138.6	McLusky and Phillips 1975
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	567800	157280.6	60	2621.3	McLusky and Phillips 1975
Copper	Polychaete (*Phyllodoce maculata*)	Invert	SW	21	WB	477500	132267.5	70	1889.5	McLusky and Phillips 1975
Copper	Polychaete (*Eudistylia vancouveri*)	Invert	SW	29	WB	6400	1280	0.14	9142.9	Young et al. 1979
Copper	Polychaete (*Eudistylia vancouveri*)	Invert	SW	29	WB	9800	1960	1.62	1209.9	Young et al. 1979
Copper	Polychaete (*Eudistylia vancouveri*)	Invert	SW	29	WB	14700	2940	3.26	901.8	Young et al. 1979
Copper	Polychaete (*Eudistylia vancouveri*)	Invert	SW	29	WB	30200	6040	6.26	964.9	Young et al. 1979
Copper	Polychaete (*Neanthes arenaceodentata*)	Invert	SW	28	WB	590	118	0.04	2950.0	Pesch and Morgan 1978
Copper	Polychaete (*Cirriformia spirabranchia*)	Invert	SW	24	WB	52000	10400	40	260.0	Millanovich et al. 1976
Copper	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	21000	2100	3	700.0	Kraak et al. 1992
Copper	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	77	Soft parts	18000	1800	13	138.5	Kraak et al. 1992
Copper	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	63	Soft parts	16000	1600	3	533.3	Kraak et al. 1992
Copper	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	63	Soft parts	200000	20000	53	377.4	Kraak et al. 1992
Copper	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	63	Soft parts	200000	20000	72	277.8	Kraak et al. 1992
Copper	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	63	Soft parts	400000	40000	90	444.4	Kraak et al. 1992
Copper	Aquatic moss (*Rhynchostegium riparioides*)	Plant	FW	27	WB	1800000	180000	21.6	8333.3	Mersch et al. 1993
Lead	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	11000	1100	31	35.5	Riley and Roth 1971
Lead	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	16500	1650	31	53.2	Riley and Roth 1971
Lead	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	8100	810	31	26.1	Riley and Roth 1971
Lead	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	56500	5650	31	182.3	Riley and Roth 1971
Lead	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	32000	3200	31	103.2	Riley and Roth 1971
Lead	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	50000	5000	31	161.3	Riley and Roth 1971
Lead	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	21200	2120	31	68.4	Riley and Roth 1971
Lead	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	9200	920	31	29.7	Riley and Roth 1971
Lead	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	46300	4630	31	149.4	Riley and Roth 1971
Lead	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	12000	1200	31	38.7	Riley and Roth 1971
Lead	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	20300	2030	31	65.5	Riley and Roth 1971
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	150000	15000	25	600.0	Canterford et al. 1978
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	100000	10000	50	200.0	Canterford et al. 1978
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	80000	8000	100	80.0	Canterford et al. 1978
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	130000	13000	150	86.7	Canterford et al. 1978
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	140000	14000	300	46.7	Canterford et al. 1978
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	150000	15000	500	30.0	Canterford et al. 1978
Lead	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	400000	40000	750	53.3	Canterford et al. 1978
Lead	Brook trout (*Salvelinus fontinalis*)	Fish	FW	90	WB	360	72	0.9	80	Holcombe et al. 1976
Lead	Brook trout (*Salvelinus fontinalis*)	Fish	FW	90	WB	8000	1600	34	47	Holcombe et al. 1976
Lead	Brook trout (*Salvelinus fontinalis*)	Fish	FW	90	WB	12700	2540	58	44	Holcombe et al. 1976
Lead	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	1300	260	0.4	650.0	Borgmann et al. 1993
Lead	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	5800	1160	3.3	351.5	Borgmann et al. 1993
Lead	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	7100	1420	2.6	546.2	Borgmann et al. 1993

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Lead	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	15800	3160	11.6	272.4	Borgmann et al. 1993
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	33000	3300	5	660.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	64000	6400	10	640.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	360000	36000	50	720.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	640000	64000	100	640.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	1400000	140000	200	700.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	2500000	250000	500	500.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	5000000	500000	1000	500.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	18000000	1800000	5000	360.0	Schulz-Baldes 1974
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	130	Soft parts	12840000	1284000	500	2568.0	Schulz-Baldes 1972
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	130	Soft parts	20770000	2077000	1000	2077.0	Schulz-Baldes 1972
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	130	Soft parts	39830000	3983000	5000	796.6	Schulz-Baldes 1972
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	112593	15200	10	1520.0	Phillips 1976
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	113333	15300	20	765.0	Phillips 1976
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	148889	20100	40	502.5	Phillips 1976
Lead	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	82222	11100	20	555.0	Phillips 1976
Lead	Caddisfly (*Brachycentrus* sp.)	Invert	FW	28	WB	300000	60000	32	1875.0	Spehar et al. 1978
Lead	Caddisfly (*Brachycentrus* sp.)	Invert	FW	28	WB	300000	60000	67	895.5	Spehar et al. 1978
Lead	Caddisfly (*Brachycentrus* sp.)	Invert	FW	28	WB	300000	60000	136	441.2	Spehar et al. 1978
Lead	Caddisfly (*Brachycentrus* sp.)	Invert	FW	28	WB	600000	120000	277	433.2	Spehar et al. 1978
Lead	Caddisfly (*Brachycentrus* sp.)	Invert	FW	28	WB	1000000	200000	565	354.0	Spehar et al. 1978
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW		Soft parts	—	—	—	536	USEPA 1985e
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	49	Soft parts	—	17000	25	680.0	Pringle et al. 1968
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	49	Soft parts	—	35000	50	700.0	Pringle et al. 1968
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	49	Soft parts	—	75000	100	750.0	Pringle et al. 1968
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	49	Soft parts	—	200000	200	1000.0	Pringle et al. 1968
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	70	Soft parts	—	35100	25	1404.0	Shuster and Pringle 1969
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	70	Soft parts	—	57590	50	1151.8	Shuster and Pringle 1969
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	70	Soft parts	—	102850	100	1028.5	Shuster and Pringle 1969
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	70	Soft parts	—	276750	200	1383.8	Shuster and Pringle 1969
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft tissue	6570	657	1	657.0	Zaroogian et al. 1979
Lead	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft tissue	11420	1142	3.3	346.1	Zaroogian et al. 1979
Lead	Hard clam (*Mercenaria mercenaria*)	Invert	SW	56	Soft parts	—	35000	200	175.0	Pringle et al. 1968
Lead	Isopod (*Asellus meridianus*)	Invert	FW	14	WB	20000000	4000000	500	8000.0	Brown 1977
Lead	Snail (*Lymnaea palustris*)	Invert	FW	120	WB	8500	2500	1	2500.0	Borgmann et al. 1978
Lead	Snail (*Physa integra*)	Invert	FW	28	WB	100000	20000	32	625.0	Spehar et al. 1978
Lead	Snail (*Physa integra*)	Invert	FW	28	WB	400000	80000	67	1194.0	Spehar et al. 1978
Lead	Snail (*Physa integra*)	Invert	FW	28	WB	500000	100000	136	735.3	Spehar et al. 1978
Lead	Snail (*Physa integra*)	Invert	FW	28	WB	500000	100000	277	361.0	Spehar et al. 1978
Lead	Snail (*Physa integra*)	Invert	FW	28	WB	1000000	200000	565	354.0	Spehar et al. 1978

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (μg/kg-dry)	Tissue (μg/kg-wet)	Water (μg/L)	BCF	Reference
Lead	Soft-shell clam (*Mya arenaria*)	Invert	SW	70	Soft parts	—	112000	100	1120.0	Pringle et al. 1968
Lead	Soft-shell clam (*Mya arenaria*)	Invert	SW	40	Soft parts	—	235000	200	1175.0	Pringle et al. 1968
Lead	Soft-shell clam (*Mya arenaria*)	Invert	SW	84	Soft parts	—	260000	200	1300.0	Pringle et al. 1968
Lead	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	300000	60000	32	1875.0	Spehar et al. 1978
Lead	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	500000	100000	67	1492.5	Spehar et al. 1978
Lead	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	500000	100000	136	735.3	Spehar et al. 1978
Lead	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	1000000	200000	277	722.0	Spehar et al. 1978
Lead	Stonefly (*Pteronarcys dorsata*)	Invert	FW	28	WB	2000000	400000	565	708.0	Spehar et al. 1978
Lead	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	900	90	0.5	180.0	Kraak et al. 1994
Lead	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	10000	1000	4	250.0	Kraak et al. 1994
Lead	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	11000	1100	10	110.0	Kraak et al. 1994
Lead	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	40000	4000	36	111.1	Kraak et al. 1994
Lead	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	130000	13000	85	152.9	Kraak et al. 1994
Manganese	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	54000	5400	5000	1.1	Riley and Roth 1971
Manganese	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	6800	680	5000	0.1	Riley and Roth 1971
Manganese	Algae (*Chlorella salina*)	Algae	SW	25	Cells	48000	4800	5000	1.0	Riley and Roth 1971
Manganese	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	11500	1150	5000	0.2	Riley and Roth 1971
Manganese	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	3800	380	5000	0.1	Riley and Roth 1971
Manganese	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	33000	3300	5000	0.7	Riley and Roth 1971
Manganese	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	6000	600	5000	0.1	Riley and Roth 1971
Manganese	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	14500	1450	5000	0.3	Riley and Roth 1971
Manganese	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	29000	2900	5000	0.6	Riley and Roth 1971
Manganese	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	69000	6900	5000	1.4	Riley and Roth 1971
Manganese	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	20000	2000	5000	0.4	Riley and Roth 1971
Manganese	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	73000	7300	5000	1.5	Riley and Roth 1971
Manganese	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	30000	3000	5000	0.6	Riley and Roth 1971
Manganese	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	4600	460	5000	0.1	Riley and Roth 1971
Manganese	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	62000	6200	5000	1.2	Riley and Roth 1971
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Bile	—	18000	100	180.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Digestive tract	—	14500	100	145.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Gill filaments	—	66900	100	669.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Kidneys	—	115700	100	1157.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Liver	—	48500	100	485.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Muscles	—	13400	100	134.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Skin	—	18800	100	188.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	Spleen	—	110000	100	1100.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Eel (*Anguilla anguilla*)	Fish	SW	32	WB	—	15300	100	153.0	Noel-Lambot and Bouquegneau 1977
Mercury (inorganic)	Fathead minnow (*Pimephales promelas*)	Fish	FW	60	WB	—	800	0.31	2580.6	Snarski and Olson 1982
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Brain	—	72000	64	1125.0	Niimi and Kissoon 1994
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Gill	—	76000	64	1187.5	Niimi and Kissoon 1994

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Kidney	—	395000	64	6171.9	Niimi and Kissoon 1994
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Liver	—	233000	64	3640.6	Niimi and Kissoon 1994
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Muscle	—	2900	64	45.3	Niimi and Kissoon 1994
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	1	Muscle, bone, skin	1700	340	560	0.6	Wilson 1983
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	1	Muscle, bone, skin	1200	240	75	3.2	Wilson 1983
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	1	Muscle, bone, skin	1700	340	1000	0.3	Wilson 1983
Mercury (inorganic)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	Spleen	—	117000	64	1828.1	Niimi and Kissoon 1994
Mercury (inorganic)	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	420	84	0.05	1680.0	Borgmann et al. 1993
Mercury (inorganic)	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	25000	5000	0.62	8064.5	Borgmann et al. 1993
Mercury (inorganic)	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	56000	11200	1.12	10000.0	Borgmann et al. 1993
Mercury (inorganic)	Eastern oyster (*Crassostrea virginica*)	Invert	SW		Soft parts	—	—	—	10,000	USEPA 1985f
Mercury (inorganic)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	60	Soft parts	—	16000	10	1,600	Cunningham and Tripp 1973
Mercury (inorganic)	Eastern oyster (*Crassostrea virginica*)	Invert	SW	60	Soft parts	—	100000	100	1,000	Cunningham and Tripp 1973
Mercury (inorganic)	Water flea (*Daphnia magna*)	Invert	FW	21	WB	1260	126	<0.01	<12600	Biesinger et al. 1982
Mercury (inorganic)	Water flea (*Daphnia magna*)	Invert	FW	21	WB	8590	859	0.36	2386	Biesinger et al. 1982
Mercury (inorganic)	Water flea (*Daphnia magna*)	Invert	FW	21	WB	15260	1526	0.72	2119	Biesinger et al. 1982
Nickel	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	7100	710	8	89	Riley and Roth 1971
Nickel	Algae (*Chlorella salina*)	Algae	SW	25	Cells	3100	310	8	39	Riley and Roth 1971
Nickel	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	6400	640	8	80	Riley and Roth 1971
Nickel	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	4300	430	8	54	Riley and Roth 1971
Nickel	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	3300	330	8	41	Riley and Roth 1971
Nickel	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	2800	280	8	35	Riley and Roth 1971
Nickel	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	10300	1030	8	129	Riley and Roth 1971
Nickel	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	6700	670	8	84	Riley and Roth 1971
Nickel	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	7500	750	8	94	Riley and Roth 1971
Nickel	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	2700	270	8	34	Riley and Roth 1971
Nickel	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	6200	620	8	78	Riley and Roth 1971
Nickel	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	4900	490	8	61	Riley and Roth 1971
Nickel	Algae (*Scenedesmus acuminata*)	Algae	FW	6	WB	—	—	1000	9.3	USEPA 1986
Nickel	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	2900	290	8	36	Riley and Roth 1971
Nickel	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	5600	560	8	70	Riley and Roth 1971
Nickel	Brown macroalgae (*Ascophylium nodosum*)	Algae	SW	Field	WB	2750	550	1.2	458.3	USEPA 1986
Nickel	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	11130	2226	21	106	Lind et al. Manuscript
Nickel	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	17600	3520	44.4	79	Lind et al. Manuscript
Nickel	Fathead minnow (*Pimephales promelas*)	Fish	FW	30	WB	25480	5096	108.9	47	Lind et al. Manuscript
Nickel	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Kidney	—	4023	1000	4.0	Calamari et al. 1982
Nickel	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Liver	—	2923	1000	2.9	Calamari et al. 1982
Nickel	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	180	Muscle	—	816	1000	0.8	Calamari et al. 1982
Nickel	Bivalve (*Cerastoderma edule*)	Invert	SW	26	WB	59600	5960	0.1	59600.0	Wilson 1983
Nickel	Bivalve (*Cerastoderma edule*)	Invert	SW	26	WB	133700	13370	1	13370.0	Wilson 1983

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Nickel	Bivalve (*Cerastoderma edule*)	Invert	SW	26	WB	319800	31980	10	3198.0	Wilson 1983
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	6000	600	4.4	136.4	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	6000	600	4.4	136.4	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	10000	1000	4.4	227.3	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	11000	1100	4.4	250.0	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	15000	1500	4.4	340.9	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	14000	1400	10	140.0	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	15000	1500	10	150.0	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	15000	1500	10	150.0	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	16000	1600	10	160.0	Zaroogian and Johnson 1984
Nickel	Blue mussel (*Mytilus edulis*)	Invert	SW	84	Soft parts	22000	2200	10	220.0	Zaroogian and Johnson 1984
Nickel	Cladoceran (*Daphnia magna*)	Invert	FW		WB	—	—	—	100	USEPA 1986
Nickel	Cladoceran (*Daphnia magna*)	Invert	FW	3.75	WB	—	9600	50	192.0	Hall 1982
Nickel	Cladoceran (*Daphnia magna*)	Invert	FW	3.75	WB	—	92250	750	123.0	Hall 1982
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	7000	700	4.2	166.7	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	8000	800	4.2	190.5	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	10000	1000	4.2	238.1	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	11000	1100	4.2	261.9	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	14000	1400	4.2	333.3	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	11000	1100	9.9	111.1	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	13000	1300	9.9	131.3	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	15000	1500	9.9	151.5	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	16000	1600	9.9	161.6	Zaroogian and Johnson 1984
Nickel	Eastern oyster (*Crassostrea virginica*)	Invert	SW	84	Soft parts	18000	1800	9.9	181.8	Zaroogian and Johnson 1984
Nickel	Rockweed (*Fucus vesiculosis*)	Plant	SW	Field	WB	4050	810	1.2	675.0	USEPA 1986
Selenium (6:1 VI:IV)	Bluegill (*Lepomis macrochirus*)	Fish	SW	60	WB	1000	200	20	10.0	Cleveland et al. 1993
Selenium (6:1 VI:IV)	Bluegill (*Lepomis macrochirus*)	Fish	SW	60	WB	2600	520	160	3.3	Cleveland et al. 1993
Selenium (6:1 VI:IV)	Bluegill (*Lepomis macrochirus*)	Fish	SW	60	WB	3800	760	330	2.3	Cleveland et al. 1993
Selenium (IV)	Bluegill (*Lepomis macrochirus*)	Fish	FW		WB	—	4500	10	450	USEPA 1987a
Selenium (IV)	Bluegill (*Lepomis macrochirus*)	Fish	FW		WB	—	4700	10	470	USEPA 1987a
Selenium (IV)	Bluegill (*Lepomis macrochirus*)	Fish	FW		WB	—	4300	10	430	USEPA 1987a
Selenium (IV)	Bluegill (*Lepomis macrochirus*)	Fish	FW		WB	—	4600	10	460	USEPA 1987a
Selenium (IV)	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	2400	120	20	Barrows et al. 1980
Selenium (IV)	Fathead minnow (*Pimephales promelas*)	Fish	FW	96	WB	—	300	11.57	25.9	Adams 1976
Selenium (IV)	Fathead minnow (*Pimephales promelas*)	Fish	FW	96	WB	—	400	24.42	16.4	Adams 1976
Selenium (IV)	Fathead minnow (*Pimephales promelas*)	Fish	FW	96	WB	—	600	50.57	11.9	Adams 1976
Selenium (IV)	Largemouth bass (*Micropterus salmoides*)	Fish	FW		WB	—	3100	10	310	USEPA 1987a
Selenium (IV)	Largemouth bass (*Micropterus salmoides*)	Fish	FW		WB	—	3000	10	300	USEPA 1987a
Selenium (IV)	Largemouth bass (*Micropterus salmoides*)	Fish	FW		WB	—	3000	10	300	USEPA 1987a
Selenium (IV)	Largemouth bass (*Micropterus salmoides*)	Fish	FW		WB	—	2700	10	270	USEPA 1987a

Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Selenium (IV)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	96	WB	—	3250	310	10.5	Adams 1976
Selenium (IV)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	96	WB	—	2460	410	6.0	Adams 1976
Selenium (IV)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	308	WB	—	160	0.3	533	Hodson et al. 1980
Selenium (IV)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	308	WB	—	440	47	9.4	Hodson et al. 1980
Selenium (IV)	Copepod (*Meganyctiphanes norvegica*)	Invert	SW	28	WB	—	—	—	200	Fowler and Benayoun 1976
Selenium (VI)	Fathead minnow (*Pimephales promelas*)	Fish	FW	56	WB	558	145.08	10.7	13.6	Bertram and Brooks 1986
Selenium (VI)	Fathead minnow (*Pimephales promelas*)	Fish	FW	56	WB	577	150.02	21.5	7.0	Bertram and Brooks 1986
Selenium (VI)	Fathead minnow (*Pimephales promelas*)	Fish	FW	56	WB	900	234	43.5	5.4	Bertram and Brooks 1986
Selenium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	WB	—	530	7.8	67.9	Hunn et al. 1987
Selenium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	WB	—	880	12.4	71.0	Hunn et al. 1987
Selenium (VI)	Rainbow trout (*Oncorhynchus mykiss*)	Fish	FW	90	WB	—	640	21	30.5	Hunn et al. 1987
Selenium (VI)	Striped bass (*Morone saxatilis*)	Fish	SW	60	WB	—	0	90	0	USEPA 1987a
Selenium (VI)	Striped bass (*Morone saxatilis*)	Fish	SW	60	WB	—	1060.2	90	11.8	USEPA 1987a
Selenium (VI)	Striped bass (*Morone saxatilis*)	Fish	SW	60	WB	—	877.2	1290	0.68	USEPA 1987a
Selenium (VI)	Striped bass (*Morone saxatilis*)	Fish	SW	60	WB	—	890.1	1290	0.69	USEPA 1987a
Silver	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	10000	1000	0.8	1250	Riley and Roth 1971
Silver	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	8800	880	0.8	1100	Riley and Roth 1971
Silver	Algae (*Chlorella salina*)	Algae	SW	25	Cells	4600	460	0.8	575	Riley and Roth 1971
Silver	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	9000	900	0.8	1125	Riley and Roth 1971
Silver	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	7800	780	0.8	975	Riley and Roth 1971
Silver	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	14100	1410	0.8	1763	Riley and Roth 1971
Silver	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	3700	370	0.8	463	Riley and Roth 1971
Silver	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	10200	1020	0.8	1275	Riley and Roth 1971
Silver	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	4500	450	0.8	563	Riley and Roth 1971
Silver	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	6200	620	0.8	775	Riley and Roth 1971
Silver	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	13700	1370	0.8	1713	Riley and Roth 1971
Silver	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	6600	660	0.8	825	Riley and Roth 1971
Silver	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	9700	970	0.8	1213	Riley and Roth 1971
Silver	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	3800	380	0.8	475	Riley and Roth 1971
Silver	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	12600	1260	0.8	1575	Riley and Roth 1971
Silver	Bluegill (*Lepomis macrochirus*)	Fish	FW	180	WB	—	150	10	15	USEPA 1987b
Silver	Bluegill (*Lepomis macrochirus*)	Fish	FW	180	WB	—	15000	100	150	USEPA 1987b
Silver	Largemouth bass (*Micropterus salmoides*)	Fish	FW	120	Muscle	—	11	1	11	USEPA 1987b
Silver	Largemouth bass (*Micropterus salmoides*)	Fish	FW	120	Muscle	—	190	10	19	USEPA 1987b
Silver	Blue mussel (*Mytilus edulis*)	Invert	SW	630	Soft parts	7650	765	1	765	Calabrese et al. 1984
Silver	Blue mussel (*Mytilus edulis*)	Invert	SW	630	Soft parts	7750	775	5	155	Calabrese et al. 1984
Silver	Blue mussel (*Mytilus edulis*)	Invert	SW	630	Soft parts	10550	1055	10	106	Calabrese et al. 1984
Silver	Gastropod (*Crepidula fornicata*)	Invert	SW	730	WB	—	10800	109.93	98	Nelson et al. 1983
Silver	Mayfly (*Ephemerella grandis*)	Invert	FW		WB	—	—	—	35	USEPA 1980
Silver	Mayfly (*Ephemerella grandis*)	Invert	FW		WB	—	—	—	240	USEPA 1980

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Table A-1. Database of bioconcentration factors (BCFs) for metals/metalloids in aquatic biota.

Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Silver	Stonefly (*Claasenia sabulosa*)	Invert	FW		WB	—	—	—	15	USEPA 1980
Silver	Stonefly (*Pteronarcys californica*)	Invert	FW		WB	—	—	—	21	USEPA 1980
Silver	Stonefly (*Pteronarcys californica*)	Invert	FW		WB	—	—	—	170	USEPA 1980
Silver	Stonefly (*Pteronarcys californica*)	Invert	FW		WB	—	—	—	79	USEPA 1980
Thallium	Atlantic salmon (*Salmo salar*)	Fish	FW	>83	Gills	—	25597	17.9	1430	Zitco et al. 1975
Thallium	Atlantic salmon (*Salmo salar*)	Fish	FW	>83	Liver	—	3866	17.9	216	Zitco et al. 1975
Thallium	Atlantic salmon (*Salmo salar*)	Fish	FW	>83	Muscle	—	2327	17.9	130	Zitco et al. 1975
Thallium	Bluegill (*Lepomis macrochirus*)	Fish	FW	28	WB	—	2720	80	34	Barrows et al. 1980
Thallium	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	2170	217	50.5	4.29703	Zitko and Carson 1975
Thallium	Blue mussel (*Mytilus edulis*)	Invert	SW	40	Soft parts	5200	520	101.5	5.123153	Zitko and Carson 1975
Thallium	Soft-shell clam (*Mya arenaria*)	Invert	SW	88	Soft parts	3780	378	47.2	8.008475	Zitko and Carson 1975
Thallium	Soft-shell clam (*Mya arenaria*)	Invert	SW	88	Soft parts	10780	1078	103.6	10.40541	Zitko and Carson 1975
Tin (inorganic)	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	35000	3500	10	350	Riley and Roth 1971
Tin (inorganic)	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	29500	2950	10	295	Riley and Roth 1971
Tin (inorganic)	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	34000	3400	10	340	Riley and Roth 1971
Tin (inorganic)	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	17500	1750	10	175	Riley and Roth 1971
Tin (inorganic)	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	45500	4550	10	455	Riley and Roth 1971
Tin (inorganic)	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	28000	2800	10	280	Riley and Roth 1971
Tin (inorganic)	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	96000	9600	10	960	Riley and Roth 1971
Tin (inorganic)	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	22400	2240	10	224	Riley and Roth 1971
Tin (inorganic)	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	49500	4950	10	495	Riley and Roth 1971
Tin (inorganic)	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	101000	10100	10	1010	Riley and Roth 1971
Tin (inorganic)	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	11000	1100	10	110	Riley and Roth 1971
Tin (inorganic)	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	34500	3450	10	345	Riley and Roth 1971
Vanadium	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	1200	120	20	6	Riley and Roth 1971
Vanadium	Algae (*Chlorella salina*)	Algae	SW	25	Cells	3700	370	20	19	Riley and Roth 1971
Vanadium	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	2400	240	20	12	Riley and Roth 1971
Vanadium	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	2900	290	20	15	Riley and Roth 1971
Vanadium	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	5700	570	20	29	Riley and Roth 1971
Vanadium	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	3100	310	20	16	Riley and Roth 1971
Vanadium	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	2400	240	20	12	Riley and Roth 1971
Zinc	Algae (*Asterionella japonica*)	Algae	SW	25	Cells	115000	11500	150	77	Riley and Roth 1971
Zinc	Algae (*Chlamydomonas* sp.)	Algae	SW	25	Cells	116000	11600	150	77	Riley and Roth 1971
Zinc	Algae (*Chlorella salina*)	Algae	SW	25	Cells	301000	30100	150	201	Riley and Roth 1971
Zinc	Algae (*Dunaliella primolecta*)	Algae	SW	25	Cells	405000	40500	150	270	Riley and Roth 1971
Zinc	Algae (*Dunaliella tertiolecta*)	Algae	SW	25	Cells	285000	28500	150	190	Riley and Roth 1971
Zinc	Algae (*Hemiselmis brunescens*)	Algae	SW	25	Cells	480000	48000	150	320	Riley and Roth 1971
Zinc	Algae (*Hemiselmis virescens*)	Algae	SW	25	Cells	259000	25900	150	173	Riley and Roth 1971
Zinc	Algae (*Heteromastix longifillis*)	Algae	SW	25	Cells	325000	32500	150	217	Riley and Roth 1971
Zinc	Algae (*Micromonas squamata*)	Algae	SW	25	Cells	105000	10500	150	70	Riley and Roth 1971

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Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Zinc	Algae (*Monochrysis lutheri*)	Algae	SW	25	Cells	160000	16000	150	107	Riley and Roth 1971
Zinc	Algae (*Olisthodiscus luteus*)	Algae	SW	25	Cells	75000	7500	150	50	Riley and Roth 1971
Zinc	Algae (*Phaeodactylum tricornutum*)	Algae	SW	25	Cells	325000	32500	150	217	Riley and Roth 1971
Zinc	Algae (*Pseudopedinella pyriformis*)	Algae	SW	25	Cells	243000	24300	150	162	Riley and Roth 1971
Zinc	Algae (*Stichococcus bacillaris*)	Algae	SW	25	Cells	251000	25100	150	167	Riley and Roth 1971
Zinc	Algae (*Tetraseimis tetrathele*)	Algae	SW	25	Cells	410000	41000	150	273	Riley and Roth 1971
Zinc	Brown macroalga (*Ascophylium nodosum*)	Algae	SW	Field	WB	—	—	11.3	1318	USEPA 1987c
Zinc	Brown macroalga (*Fucus serratus*)	Algae	SW	140	WB	—	—	9.5	10768	USEPA 1987c
Zinc	Brown macroalga (*Fucus serratus*)	Algae	SW	Field	WB	—	—	5.21-11.9	2029	USEPA 1987c
Zinc	Brown macroalga (*Fucus serratus*)	Algae	SW	Field	WB	—	—	11.3	1027	USEPA 1987c
Zinc	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	120000	12000	25	480.0	Canterford et al. 1978
Zinc	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	100000	10000	50	200.0	Canterford et al. 1978
Zinc	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	140000	14000	100	140.0	Canterford et al. 1978
Zinc	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	170000	17000	150	113.3	Canterford et al. 1978
Zinc	Diatom (*Ditylum brightwellii*)	Algae	SW	14	Cells	150000	15000	200	75.0	Canterford et al. 1978
Zinc	Algae (*Dunaliella tertiolecta*)	Algae	SW	0.5	WB	—	—	7.2-98000	10000	USEPA 1987c
Zinc	Diatom (*Thalassiosira pseudonana*)	Algae	SW	0.5	WB	—	—	7.2-98000	12000	USEPA 1987c
Zinc	Atlantic salmon (*Salmo salar*)	Fish	FW	80	WB	—	28000	2	14000	Farmer et al. 1979
Zinc	Atlantic salmon (*Salmo salar*)	Fish	FW	80	WB	—	34000	320	106.25	Farmer et al. 1979
Zinc	Atlantic salmon (*Salmo salar*)	Fish	FW	80	WB	—	37000	560	66	Farmer et al. 1979
Zinc	Atlantic salmon (*Salmo salar*)	Fish	FW	80	WB	—	42000	742	57	Farmer et al. 1979
Zinc	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	120000	24000	4.1	5853.7	Spehar et al. 1978
Zinc	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	125000	25000	4.5	5555.6	Spehar et al. 1978
Zinc	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	320000	64000	73.4	871.9	Spehar et al. 1978
Zinc	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	270000	54000	105	514.3	Spehar et al. 1978
Zinc	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	250000	50000	127	393.7	Spehar et al. 1978
Zinc	Flagfish (*Jordanella floridae*)	Fish	FW	100	WB	270000	54000	139	388.5	Spehar et al. 1978
Zinc	Guppy (*Poecilia reticulata*)	Fish	FW	134	WB	444000	88800	166	535	Pierson 1981
Zinc	Guppy (*Poecilia reticulata*)	Fish	FW	134	WB	430000	86000	180	478	Pierson 1981
Zinc	Guppy (*Poecilia reticulata*)	Fish	FW	134	WB	828000	165600	336	493	Pierson 1981
Zinc	Guppy (*Poecilia reticulata*)	Fish	FW	134	WB	1540000	308000	319	966	Pierson 1981
Zinc	Guppy (*Poecilia reticulata*)	Fish	FW	134	WB	1550000	310000	605	512	Pierson 1981
Zinc	Guppy (*Poecilia reticulata*)	Fish	FW	134	WB	1420000	284000	609	466	Pierson 1981
Zinc	Mummichog (*Fundulus heteroclitus*)	Fish	SW	56	WB	201000	40200	210	191.4	Sauer and Watabe 1984
Zinc	Mummichog (*Fundulus heteroclitus*)	Fish	SW	56	WB	252000	50400	810	62.2	Sauer and Watabe 1984
Zinc	Mummichog (*Fundulus heteroclitus*)	Fish	SW	56	WB	413000	82600	7880	10.5	Sauer and Watabe 1984
Zinc	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	125000	25000	58	431.0	Ahsanulla and Williams 1991
Zinc	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	130000	26000	82	317.1	Ahsanulla and Williams 1991
Zinc	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	145000	29000	131	221.4	Ahsanulla and Williams 1991
Zinc	Amphipod (*Allorchestes compressa*)	Invert	SW	28	WB	160000	32000	140	228.6	Ahsanulla and Williams 1991

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Metal/Metalloid	Species	Organism Type	Water Type	Duration (days)	Tissue	Tissue (µg/kg-dry)	Tissue (µg/kg-wet)	Water (µg/L)	BCF	Reference
Zinc	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	74000	14800	5.6	2642.9	Borgmann et al. 1993
Zinc	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	66000	13200	13	1015.4	Borgmann et al. 1993
Zinc	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	85000	17000	21.2	801.9	Borgmann et al. 1993
Zinc	Amphipod (*Hyalella azteca*)	Invert	FW	70	WB	126000	25200	42.3	595.7	Borgmann et al. 1993
Zinc	Amphipod (*Hyalella azteca*)	Invert	FW	28	WB	163000	32600	301	108.3	Borgmann and Norwood 1995
Zinc	Amphipod (*Hyalella azteca*)	Invert	FW	42	WB	196000	39200	301	130.2	Borgmann and Norwood 1995
Zinc	Asiatic clam (*Corbicula fluminea*)	Invert	FW	28	Soft parts	229900	22990	218	105.5	Graney et al. 1983
Zinc	Asiatic clam (*Corbicula fluminea*)	Invert	FW	28	Soft parts	239200	23920	433	55.2	Graney et al. 1983
Zinc	Asiatic clam (*Corbicula fluminea*)	Invert	FW	28	Soft parts	289100	28910	835	34.6	Graney et al. 1983
Zinc	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	384444	51900	100	519	Phillips 1976
Zinc	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	465926	62900	200	315	Phillips 1976
Zinc	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	387407	52300	200	262	Phillips 1976
Zinc	Blue mussel (*Mytilus edulis*)	Invert	SW	35	Soft parts	545926	73700	400	184	Phillips 1976
Zinc	Soft-shell clam (*Mya arenaria*)	Invert	SW	49	Soft parts	—	27000	200	135	USEPA 1987c
Zinc	Crayfish (*Oronectes virilis*)	Invert	FW	14	Abdominal muscle	89000	17800	12200	1	Mirenda 1986a
Zinc	Crayfish (*Oronectes virilis*)	Invert	FW	14	Carapace	36100	7220	12200	1	Mirenda 1986a
Zinc	Crayfish (*Oronectes virilis*)	Invert	FW	14	Gill	333300	66660	12200	5	Mirenda 1986a
Zinc	Crayfish (*Oronectes virilis*)	Invert	FW	14	Hepatopancreas	212600	42520	12200	3	Mirenda 1986a
Zinc	Crayfish (*Oronectes virilis*)	Invert	FW	14	Intestine	100900	20180	12200	2	Mirenda 1986a
Zinc	Crayfish (*Oronectes virilis*)	Invert	FW	14	WB	63500	12700	12200	1	Mirenda 1986a
Zinc	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	2708000	100	27,080	Shuster and Pringle 1969
Zinc	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	2560000	100	25,600	Shuster and Pringle 1969
Zinc	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	3813000	200	19,065	Shuster and Pringle 1969
Zinc	Eastern oyster (*Crassostrea virginica*)	Invert	SW	140	Soft parts	—	3185000	200	15,925	Shuster and Pringle 1969
Zinc	Midge (*Chironomus riparius*)	Invert	FW	77	WB	—	2620000	1000	2620	Timmermans et al. 1992
Zinc	Periwinkle (*Littorina obtusata*)	Invert	SW		Soft parts	—	—	11	670	USEPA 1987c
Zinc	Shrimp (*Pandalus montagui*)	Invert	SW	14	WB	70000	14000	65	215	Ray et al. 1980
Zinc	Soft-shell clam (*Mya arenaria*)	Invert	SW	50	Soft parts	—	17000	200	85	Pringle et al. 1968
Zinc	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	100000	10000	3	3333.3	Kraak et al. 1994
Zinc	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	100000	10000	3	3333.3	Kraak et al. 1994
Zinc	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	200000	20000	38	526.3	Kraak et al. 1994
Zinc	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	200000	20000	101	198.0	Kraak et al. 1994
Zinc	Zebra mussel (*Dreissena polymorpha*)	Invert	FW	70	Soft parts	700000	70000	382	183.2	Kraak et al. 1994

FW = Freshwater
SW = Saltwater
WB = Whole body
— = Not reported